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Operational validation and intercomparison of different types of hydrological models

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Abstract. A theoretical framework for model validation, based on the methodology originally proposed by *Klemes* [1985, 1986], is presented. It includes a hierarchial validation testing scheme for model application to runoff prediction in gauged and ungauged catchments subject to stationary and nonstationary climate conditions. A case study on validation and intercomparison of three different models on three catchments in Zimbabwe is described. The three models represent a lumped conceptual modeling system (NAM), a distributed physically based system (MIKE SHE), and an intermediate approach (WATBAL). It is concluded that all models performed equally well when at least 1 year's data were available for calibration, while the distributed models performed marginally better for cases where no calibration was allowed.

Introduction

In recent years water resources studies have become increasingly concerned with aspects of water resources for which data are not directly available. Examples include studies of the development potential of ungauged areas, environmental impacts of land use changes related to agricultural and forestry practices, conjunctive use of groundwater and surface water, and climate impact studies concerned with the effects on water resources of an anticipated climate change.

In these and other types of studies, hydrological simulation models are often used to provide the missing information as a basis for decisions regarding the development and management of water and land resources.

Traditionally, hydrological simulation modeling systems are classified in three main groups, namely, (1) empirical black box, (2) lumped conceptual, and (3) distributed physically based systems. The great majority of the modeling systems used in practice today belongs to the simple types (1) or (2) and require a modest numbers of parameters (approximately 5–10) to be calibrated for their operation. Despite their simplicity, many models have proven quite successful in representing an already measured hydrograph.

A severe drawback of these traditional modeling systems, however, is that their parameters are not directly related to the physical conditions of the catchment. Accordingly, it may be expected that their applicability is limited to areas where runoff has been measured for some years and where no significant change in catchment conditions have occurred.

To provide a more appropriate tool for the type of studies mentioned above, considerable efforts within hydrological research have been directed toward development of distributed physically based catchment models. Such models use parameters which are related directly to the physical characteristics of the catchment (topography, soil, vegetation, and geology) and operate within a distributed framework to account for the spatial variability of both physical characteristics and meteorological conditions. These models aim at describing the hydrological processes and their interaction as and where they occur in the catchment and therefore offer the prospect of remedying the shortcomings of the traditional rainfall runoff models.

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Although there appears to be a certain degree of consensus at the theoretical level regarding the potential of the distributed physically based types of models, there are widely divergent points of view as to whether they offer a significant improvement in actual performance when compared to the well-proven lumped conceptual model type. *Beven* [1989, p. 161] argues from theoretical considerations of scale problems that "the current generation of distributed physically based models are lumped conceptual models," and, further, that all current physically based models "are not well suited to applications to real catchments." *Grayson et al.* [1992] support this view and claim that physically based models have been oversold by their developers. Other authors, for example, *Smith et al.* [1994], argue that this criticism is "overly pessimistic."

An evaluation of the capabilities of hydrological models when applied in the absence of site calibration data and limited validation data to predict the effects of major land use changes was made by the Task Committee on Quantifying Land-Use Change Effects [*U.S. Committee*, 1985], which reported a great belief among committee members in the capabilities of 28 surface water hydrological modeling systems, most of which can be classified as lumped conceptual models. In view of the limited number of model comparison studies conducted and the less-than-encouraging results often obtained, this confidence is remarkable. According to the *U.S. Committee* [1985, p. 1], "the reasons for this confidence were explored and appear to be based upon personal experience, possibly tempered by belief in the model originators."

Owing to the complexity of the problems involved, further theoretical evaluation is not likely to provide a definite conclusion regarding the capability and limitation of distributed, physically based modeling systems. For establishing a basis to better advance the discussion, relevant model validations appear to be a more fruitful approach, where the models concerned simply are subjected to a range of practical modeling tests to validate their capability for undertaking particular tasks.

In this respect, *Klemes* [1986, p. 17], has developed a hierarchial scheme for model testing, which is based on the philosophy that "a hydrological simulation model must demonstrate, before it is used operationally, how well it can perform the kind of task for which it is intended." It may appear needless to advocate such a basic and evident requirement. Unfortunately, it is well justified in view of the current practice in hydrological model testing.

The present paper is based on results from a research project conducted at the *Danish Hydraulic Institute* (*DHI*) [1993a]. The project had two major objectives. The first objective was to identify a rigorous framework for the testing of model capabilities for different types of tasks. The second objective was to use this theoretical framework and conduct an intercomparison study involving application of three modeling systems of different complexity to a number of tasks ranging from traditional simulation of stationary, gauged catchments to simulation of ungauged catchments and of catchments with nonstationary climate conditions. Data from three catchments in Zimbabwe were used for the tests. The research project was a contribution to project D.5, "Testing the transferability of hydrological simulation models," forming part of the World Climate Programme—Water [World Meteorological Organization (WMO), 1985].

Some of the results of *DHI* [1993a] were presented by *Refsgaard* [1996] with a focus on modeling the land surface processes and the coupling between hydrological and atmospheric models within the global change context. Thus *Refsgaard* [1996] presents some of the results from two of the Zimbabwean catchments to illustrate data requirements and form the basis for conclusions regarding which type of hydrological model is required for climate change modeling. The present paper, on the other hand, emphasizes the modeling methodology and contains a summary of all the test results from all the three Zimbabwian catchments. It furthermore provides a general discussion of these results with references to similar studies reported in literature.

Theoretical Framework for Model Validation

Terminology

No unique and generally accepted terminology is presently used in the hydrological community with regard to issues related to model validation. The framework used in the present paper is basically in line with the terminology defined by *Schlesinger et al.* [1979], *Tsang* [1991], and *Flavelle* [1992] and comprises the following key definitions.

A modeling system (i.e., code) is a generalized software package, which can be used for different catchments without modifying the source code. Examples of modeling systems are MIKE SHE, SACRAMENTO, and MODFLOW.

A model is a site-specific application of a modeling system, including given input data and specific parameter values. An example of a model is a MIKE SHE–based model for the Ngezi catchment (cf. the case study below).

A modeling system or a code can be "verified." A code verification involves comparison of the numerical solution generated by the code with one or more analytical solutions or with other numerical solutions. Verification ensures that the computer program accurately solves the equations that constitute the mathematical model.

Model validation is here defined as the process of demonstrating that a given site-specific model is capable of making accurate predictions for periods outside a calibration period. A model is said to be validated if its accuracy and predictive capability in the validation period have been proven to lie within acceptable limits or errors. It is important to notice that the term model validation refers to a site specific validation of a model. This must not be confused with a more general validation of a generalized modeling system which, in principle, will never be possible.

Testing Scheme for Validation of Hydrological Models

The hierarchial testing scheme proposed by *Klemes* [1985, 1986] appears suitable for testing the capability of a model to predict the hydrological effect of climate change, land use change, and other nonstationary conditions. Klemes distinguished between simulations conducted for the same station (catchment) used for calibration and simulations conducted for ungauged catchments. He also distinguished between cases where climate, land use, and other catchment characteristics remain unchanged (are stationary) and cases where they are not. This leads to the definitions of four basic categories of typical modeling tests.

1. The split-sample test (SS) involves calibration of a model based on 3–5 years of data and validation on another period of a similar length.

2. The differential split-sample test (DSS) involves calibration of a model based on data before catchment change occurs, adjustment of model parameters to characterize the change, and validation on the subsequent period.

3. In the proxy-basin test (PB) no direct calibration is allowed, but advantage may be taken of information from other gauged catchments. Hence validation will comprise identification of a gauged catchment deemed to be of a nature similar to that of the validation catchment; initial calibration; transfer of model, including adjustment of parameters to reflect actual conditions within validation catchment; and validation.

4. With the proxy-basin differential split-sample test (PB-DSS), again no direct calibration is allowed, but information from other catchments may be used. Hence validation will comprise initial calibration on the other relevant catchment, transfer of model to validation catchment, selection of two parameter sets to represent the periods before and after the change, and subsequent validations on both periods.

Relevant Literature on Model Intercomparison Studies

The testing of hydrological models through validation on independent data has for a long time been emphasized by the World Meteorological Organization (WMO). In their pioneering studies [*WMO*, 1975, 1986, 1992] several hydrological modeling systems of the empirical black box and the lumped conceptual types were tested on the same data from different catchments. The actual testing, however, only included the standard SS test comprising an initial calibration of a model and subsequent validation based on data from an independent period. No firm conclusions were derived regarding significant differences in performance among different model types.

Franchini and Pacciani [1991] made a comparative analysis of seven different lumped conceptual models. They used an SS testing approach calibrating on a 1-month period and validating on a subsequent 3-month period. They concluded that in spite of a wide range of structural complexity all the models produced similar and equally valid results. With regard to the question of whether the simpler or the more complex variants within this group of models are better, they concluded that significantly different models produced basically equivalent results, with calibration times being generally proportional to the complexity of their structure. On the other hand, they concluded that the model structure should not be made too simple, because it will then cause a loss of the link with the physics of the problem and of the possibility of taking advantage of prior knowledge of the geomorphological nature of the catchment.

Other researchers have conducted similar intercomparison studies involving empirical black box models and lumped conceptual models [*Naef*, 1981; *Wilcox et al.*, 1990] with similar conclusions.

Only a few studies have included comparisons of distributed physically based models with simpler models. *Loague and Freeze* [1985] in a classical study compared two empirical black box modeling systems (a regression model and a unit hydrograph model) and a quasi physically based system on three small experimental catchments ranging from 10 ha to 7.2 km^2. The models were used on an event basis to simulate runoff peaks. The two empirical models were calibrated against runoff data and subsequently validated on independent data in an SS approach. The parameter values for the quasi physically based model were assessed directly from field data and not subject to any calibration before being validated against the same data as the two other models. *Loague and Freeze* [1985] found that all models performed poorly. For one catchment the quasi physically based model was subsequently applied with and without calibration of one key model parameter. Such calibration had little impact on the model performance during the validation period.

In a study in the semiarid 150 km^2 Walnut Gulch experimental watershed *Michaud and Sorooshian* [1994] compared a lumped conceptual model (SCS), a distributed conceptual model (SCS with eight subcatchments, one per raingauge) and a distributed physically based model (KINEROS) for simulation of storm events. They found that with calibration, the accuracies of the two distributed models were similar. Without calibration the distributed physically based model performed better than the distributed conceptual model, and in both cases the lumped conceptual model performed poorly.

Thus, as far as the test experience for distributed physically based models is concerned, both *Loague and Freeze* [1985] and *Michaud and Sorooshian* [1994] have performed tests on relatively small experimental catchments with very good data coverage. Both studies have used the models on ungauged conditions (without calibration) but in all cases under stationary climate conditions. The present paper presents results from larger catchments in Zimbabwe with ordinary data coverage and performs a sequence of rigorous tests of increasing complexity according to the hierarchial scheme outlined by *Klemes* [1986], involving intercomparisons between lumped conceptual and distributed physically based models.

Hydrological Modeling Systems

The following three modeling systems (codes) are used in the present study: a lumped conceptual rainfall-runoff modeling system (NAM), a semidistributed hydrological modeling system (WATBAL), and a distributed physically based hydrological modeling system (MIKE SHE). The NAM and MIKE SHE can be characterized as very typical of their respective classes, while the WATBAL falls in between these two standard classes. All three modeling systems are being used on a routine basis at the Danish Hydraulic Institute (DHI) in connection with consultancy and research projects.

NAM

NAM is a traditional hydrological modeling system of the lumped conceptual type operating by continuously accounting for the moisture contents in four mutually interrelated storages. The NAM was originally developed at the Technical University of Denmark [*Nielsen and Hansen*, 1973] and has been modified and extensively applied by DHI in a large number of engineering projects covering all climatic regimes of the world. Furthermore, the NAM has been transferred to more than 100 other organizations worldwide as part of DHI's MIKE 11 generalized river modeling package. The structure of NAM is illustrated in Figure 1. The NAM has in its present version a total of 17 parameters; however, in most cases only about 10 of these are adjusted during calibration.

WATBAL

WATBAL was developed in the early 1980s by DHI in an attempt to enable full utilization of readily available, distributed data on land surface properties (topography, vegetation, and soil) in a physically based model, and yet it is simple enough to allow large-scale applications within reasonable computational requirements. Here the WATBAL is briefly introduced; more detailed information has been given by *Knudsen et al.* [1986].

WATBAL has been designed to account for the spatial and temporal variations of soil moisture. On the basis of distributed information on meteorological conditions, topography, vegetation, and soil types, the catchment area is divided into a number of hydrological response units, as illustrated in Figure 2, with each unit being characterized by a different composition of the above features. These units are used to provide the spatial representation of soil moisture, while temporal variations within each unit are accounted for by means of empirical relations for the processes affecting soil moisture, using physical parameters particular to each unit.

For the representation of subsurface flows a simple lumped, conceptual approach is applied, using a cascade of linear reservoirs to account for the interflow and baseflow components (Figure 3). In summary, WATBAL provides a distributed physically based description of the surface processes affecting soil moisture (interception, infiltration, evapotranspiration, and percolation), while a lumped conceptual approach is used to represent subsurface flows. WATBAL has previously been used successfully for prediction of runoff from ungauged catchments [*Nielsen and Bari*, 1988].

MIKE SHE

MIKE SHE is a further development of the European Hydrological System—SHE [*Abbott et al.*, 1986a, b]. It is a deterministic, fully distributed and physically based modeling system for describing the major flow processes of the entire land phase of the hydrological cycle. MIKE SHE solves the partial differential equations for the processes of overland and channel flow and unsaturated and saturated subsurface flow. The system is completed by a description of the processes of snow melt, interception, and evapotranspiration. The flow equations are solved numerically using finite difference methods.

In the horizontal plane the catchment is discretized in a

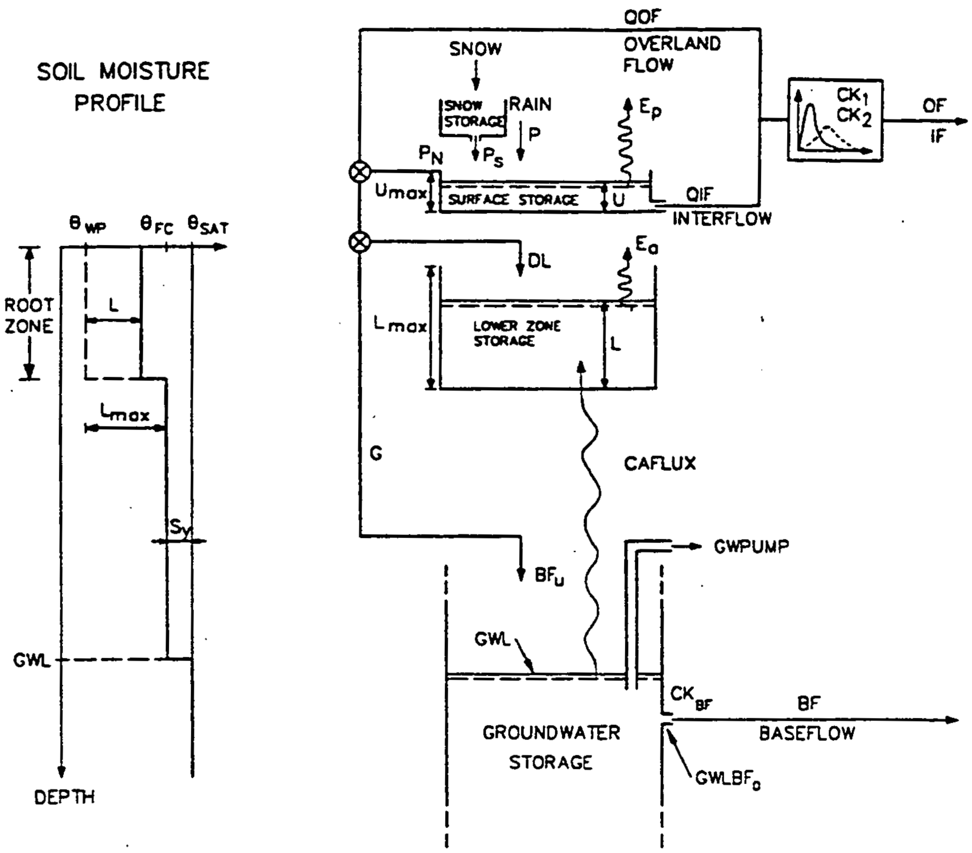

Figure 1. Structure of the NAM rainfall runoff modeling system [*DHI*, 1994].

network of grid squares. The river system is assumed to run along the boundaries of these. Within each square the soil profile is represented by a number of computational nodes in the vertical direction, which above the groundwater table may become partly saturated. Lateral subsurface flow is only considered in the saturated part of the profile. Figure 4 illustrates the structure of the MIKE SHE. A description of the methodology and some experiences of model application to ordi-

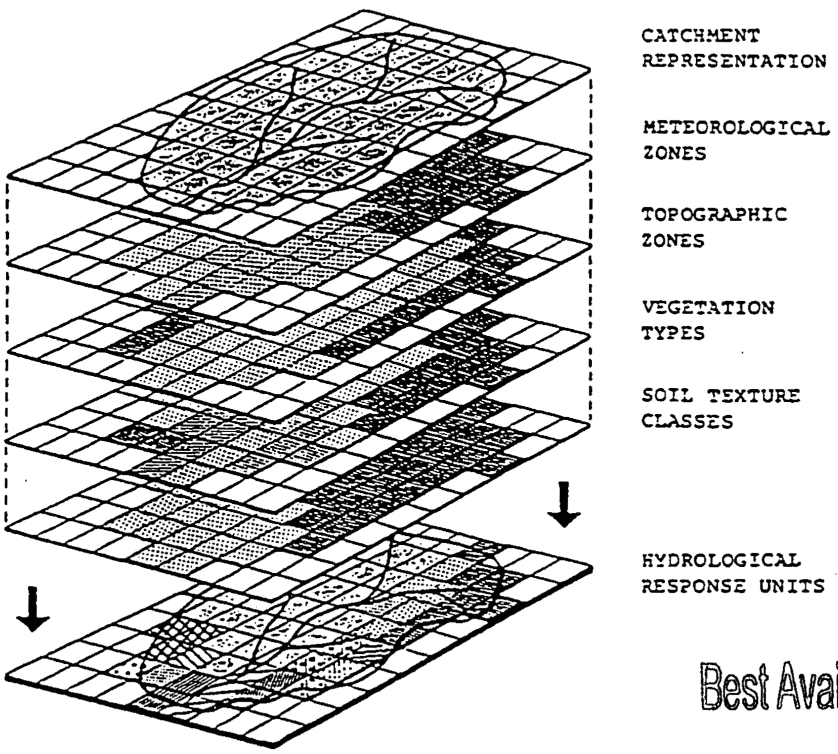

Figure 2. WATBAL representation of catchment characteristics and definition of hydrological response units [*Knudsen et al.*, 1986].

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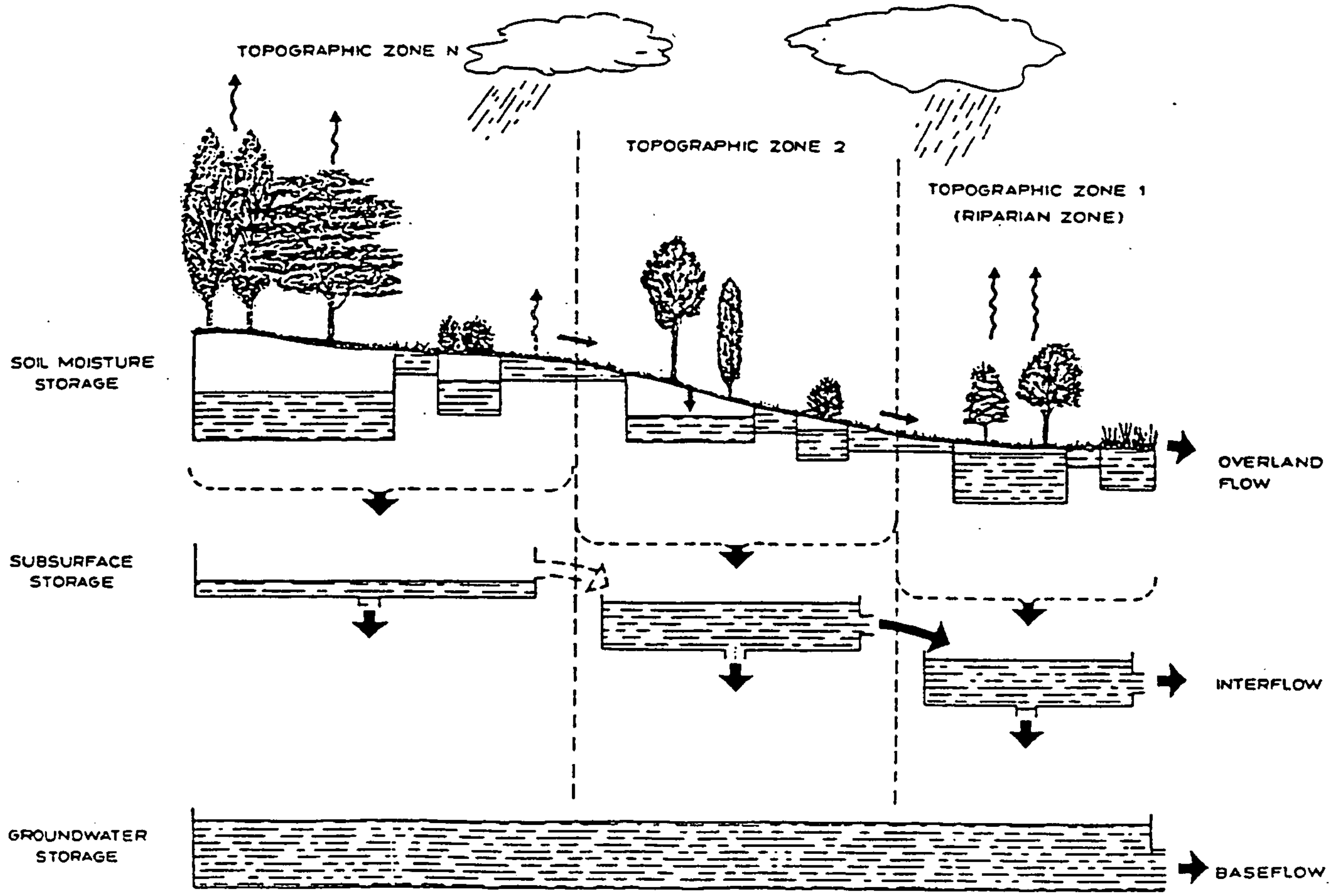

Figure 3. Principal structure of WATBAL [*Knudsen et al.*, 1986].

nary catchments have been given by *Refsgaard et al.* [1992] and *Jain et al.* [1992]. A more detailed description has been given by *Refsgaard and Storm* [1995].

MIKE SHE is usually categorized as a physically based system. The characterization is, strictly speaking, correct only if it is applied on an appropriate scale. A number of scale problems arise when the MIKE SHE is used on a regional scale [*Refsgaard and Storm*, 1995]. In addition, if there is a considerable

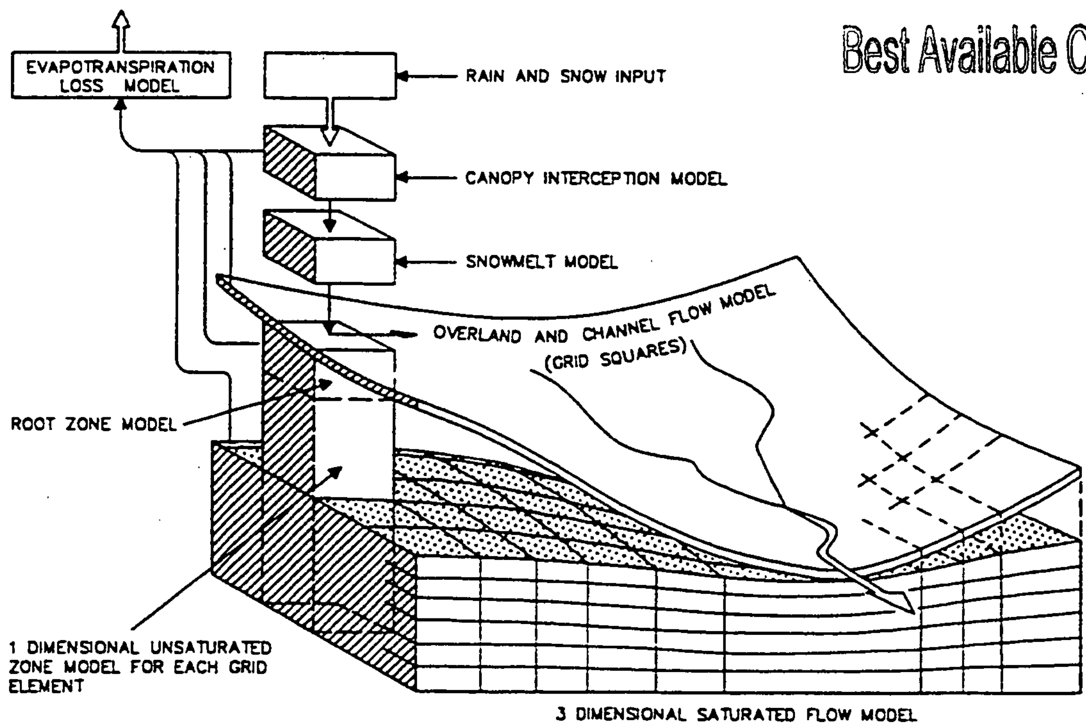

Figure 4. Schematic presentation of the MIKE SHE [*DHI*, 1993b].

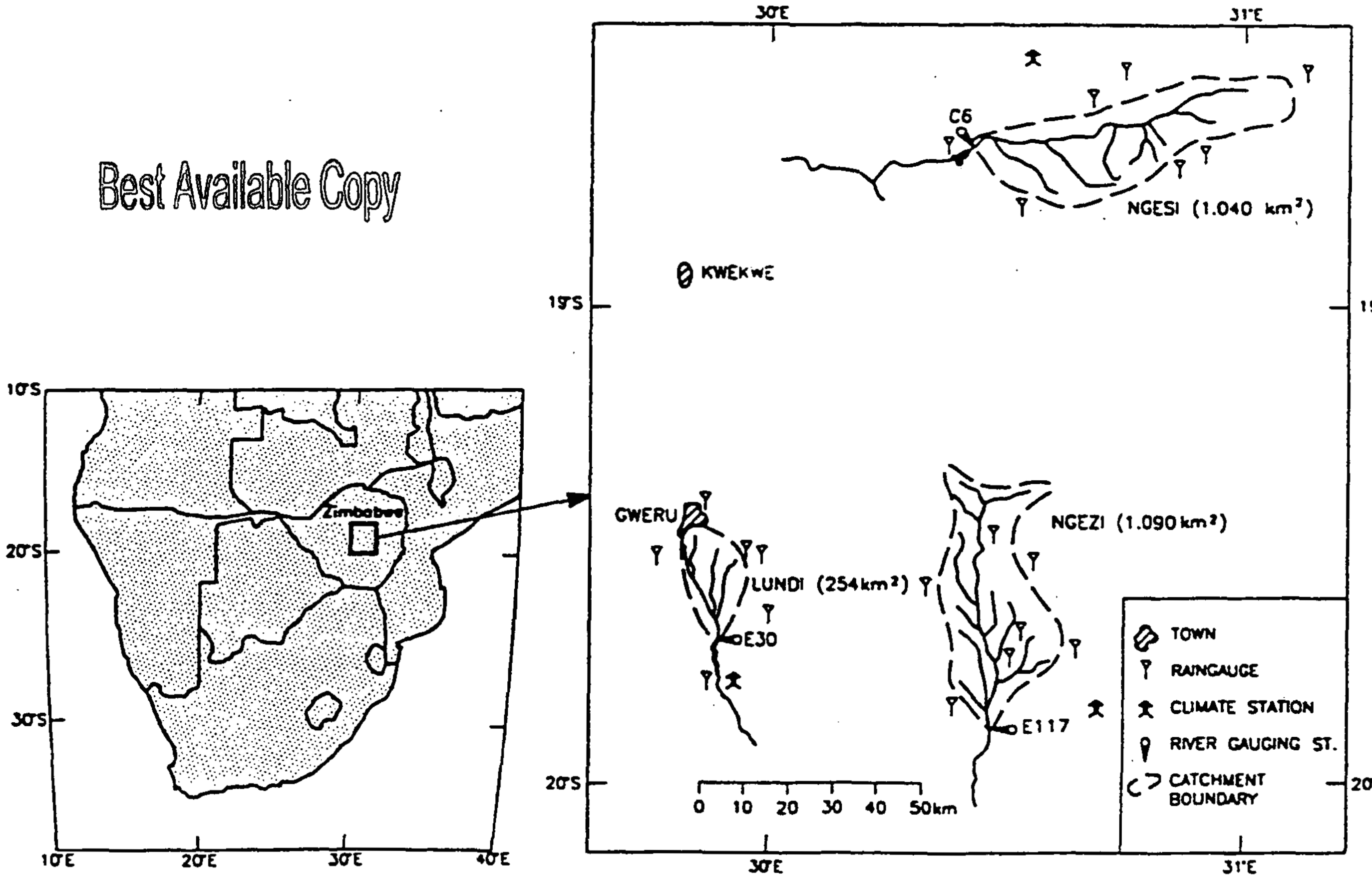

Figure 5. Location of the three catchments in Zimbabwe.

uncertainty attached to the basic information, and if the spatial and temporal variables (such as groundwater table elevations) cannot be validated against observations, a MIKE SHE model of that particular site cannot be considered fully physically based but will degenerate towards a detailed conceptual model. In this case the calibration procedure is usually to adjust the parameters with the largest uncertainties attached, within a reasonable range.

Case Study: Methodology

Selected Catchments in Zimbabwe

The three catchments in Zimbabwe that were selected for the model tests are Ngezi-South (1090 km^2), Lundi (254 km^2), and Ngezi-North (1040 km^2). The locations of the catchments are shown in Figure 5.

A brief data collection/field reconnaissance to Zimbabwe was arranged to obtain relevant information. Daily series of rainfall and monthly series of pan evaporation were obtained from the Department of Meteorological Services. Records of mean daily discharges as well as information on water rights were obtained from the Hydrological Branch, Ministry of Energy Water Resources and Development. Detailed information on land use was obtained through subcontracting R. Whitlow, University of Zimbabwe, to prepare land-use maps based upon 1:25,000 aerial photographs. Furthermore, 1:50,000 topographical maps were collected and digitized. Information on vegetation characteristics was obtained from *Timberlake* [1989] as well as from J. Timberlake and N. Nobanda, National Herbarium (personal communication, 1989); B. Campell, Department of Biological Sciences (personal communication, 1989); and G. MacLaureen, Department of Crop Science, University of Zimbabwe (personal communication, 1989). Information on soil characteristics and hydrogeology was obtained from *Anderson* [1989]. Finally, valuable information of various kinds was provided by R. Whitlow, Department of Geography, University of Zimbabwe (personal communication, 1989); H. Elwell, Agritex (personal communication, 1989); J. Anderson, Chemistry and Soil Research Institute, Ministry of Agriculture (personal communication, 1989); and others. A more detailed description is given in *DHI* [1993a].

The annual catchment rainfall and runoff for the periods selected for modeling are shown in Table 1, while some of the key features for the three catchments are presented in Table 2. It is noticed from the rainfall and runoff figures in Table 1 that there are very large interannual variations. From Table 2 it appears that there are significant differences in the vegetation and soil characteristics from catchment to catchment.

Model Testing Scheme

The model testing scheme is illustrated in Figure 6. The testing of the involved models has been undertaken in parallel and in the following sequence.

1. The SS test was based on data from Ngezi-South comprising an initial calibration of the models and a subsequent validation using data for an independent period.

2. The PB test involved transfer of models to the Lundi catchment and adjustment of parameters to reflect the prevailing catchment characteristics and validation without any calibration.

3. The modified proxy-basin (M-PB) test was as above, but

Table 1. Annual Rainfall and Runoff Values for the Three Zimbabwean Test Catchments

Hydrological Year	Rainfall, mm/yr	Runoff, mm/yr
	Ngezi-South	
1971/1972	890	131
1972/1973	317	2
1973/1974	1290	349
1974/1975	1087	236
1975/1976	879	90
1976/1977	872	116
1977/1978	1131	245
1978/1979	609	59
	Lundi	
1971/1972	920	89
1972/1973	371	2
1973/1974	1384	460
1974/1974	1046	217
1975/1976	857	89
1981/1982	416	10
1982/1983	528	7
1983/1984	547	8
	Ngezi-North	
1977/1978	1047	156
1978/1979	730	64
1981/1982	430	12
1982/1983	395	1
1983/1984	436	4

was adjusted by allowing model calibration based on 1 year of runoff data.

4. For the DSS test, model calibration was based on data from an initial calibration period, and validation was based on data from a subsequent period. The differential nature of this test is justified by the fact that the later independent period includes three successive years (1981/1982–1983/1984) with a markedly lower rainfall than would be otherwise and hence represents a nonstationary climate scenario.

5. The PB-DSS test involved transferring the models to the Ngezi-North catchment, adjusting the parameters to represent the catchment characteristics, and validating them by runoff simulation over a nonstationary climate period.

6. The modified proxy-basin differential split-sample (M-PB-DSS) test was as above, though it allowed models to be calibrated using a short-term (1 year) record.

Evaluation Criteria

For measuring the performance of the models for each test, a standard set of criteria has been defined. The criteria have been designed with the sole purpose of measuring how closely the simulated series of daily flows agree with the measured series. Owing to the generalized nature of the defined model validations, it has been necessary to introduce several criteria for measuring the performance with regard to water balance, low flows, and peak flows.

The standard set of performance criteria comprises a combination of the following four graphical plots and three numerical measures: (1) joint plots of the simulated and observed hydrographs; (2) scatter diagram of monthly runoffs; (3) flow duration curves; (4) scatter diagram of annual maximum discharges; (5) overall water balance; (6) the Nash-Sutcliffe coefficient ($R2$); and (7) an index (EI) measuring the agreement between the simulated and observed flow duration curves.

The coefficient $R2$, introduced by *Nash and Sutcliffe* [1970], is computed on the basis of the sequence of observed and simulated monthly flows over the whole testing period (perfect agreement for $R2$ is 1):

$$R2 = 1 - \sum_{m=1}^{M} (Q_m^o - Q_m^s)^2 \Big/ \sum_{m=1}^{M} (Q_m^o - \bar{Q}^o)^2$$

where

M total number of months;
Q_m^s simulated monthly flows;
Q_m^o observed monthly flows;
$\bar{Q}^o$ average observed monthly flows over whole period.

The flow duration curve error index, EI, provides a numerical measure of the difference between the flow duration curves of simulated and observed daily flows (perfect agreement for EI is 1):

$$\mathrm{EI} = 1 - \int [f_o(q) - f_s(q)]\, dq \Big/ \int f_o(q)\, dq$$

where $f_o(q)$ is the flow duration curve based on observed daily flows, and $f_s(q)$ is the flow duration curve based on simulated daily flows.

Table 2. Land-Use Vegetation and Soil Characteristics Estimated From Available Information and a Brief Field Visit

	Catchment		
	Ngezi-South	Lundi	Ngezi-North
Land use/vegetation (area %)			
Dense/closed woody vegetation	7	13	10
Open woody vegetation	36	25	35
Sparse woody vegetation	14	19	14
Grassland	11	39	16
Cropland	29	3	19
Abandoned cropland	2	0	6
Rock outcrops	1	0	0
Soil depth range, m	0–2.5	0–1	0.5–6
Saturated hydraulic conductivity in root zone soil, mm/hr	range: 1–250 average: 80	range: 1–70 average: 60	range: 2–100 average: 50
Available water content in root zone soil, vol %	range: 10–14 average: 12	range: 10–12 average: 11	range: 9–29 average: 17

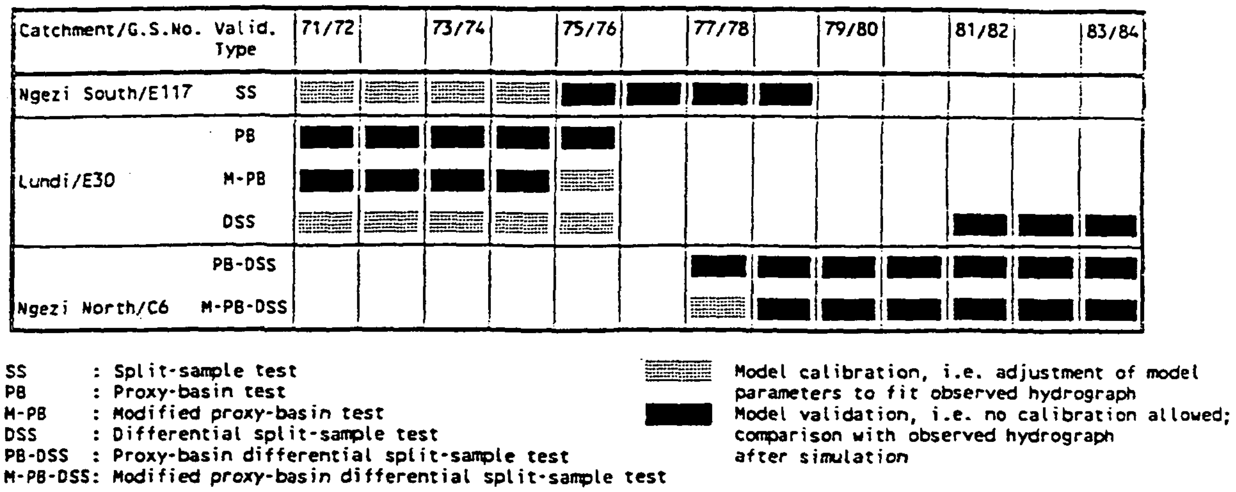

Figure 6. Model validation test schemes.

Model Construction, Calibration, and Application

All models have had access to the same hydrometeorological data and catchment information at any time. Due to the nature of the different models, however, the WATBAL and SHE have been able to make more direct use of the available information than the NAM.

In this respect, the NAM has disregarded the spatial variation of rainfall and used the catchment average series as input, and for the simulation of ungauged catchments, a subjective evaluation of catchment characteristics has been undertaken for estimation of the appropriate model parameters. On the other hand, the WATBAL and SHE have attempted to account for the spatial variability of rainfalls as well as information on typical storm durations to convert daily rainfall series to realistic hourly rainfalls. Furthermore, these models have directly used the available information on the spatial variation of topography and soil and vegetation types and their characteristics for model setup and estimation of appropriate model parameters.

As an illustration of the differences in model complexity and the different abilities of the three modeling systems to utilize the available distributed catchment data, some key facts for the three model applications to the 1090 km^2 Ngezi-South catchment are given in the following three paragraphs.

The NAM model considered the entire catchment as one unit, utilized only catchment areal rainfall, and initially disregarded information on soil, vegetation, and geology. Such information was subsequently used on a subjective basis for assessing likely parameter values in the PB tests on the other two catchments. During the model calibrations (when allowed) the values of the 10 parameters were assessed.

The WATBAL model was established on the basis of six meteorological zones, eight soil types, and 11 vegetation types. The spatial occurrences of these three features resulted in 129 hydrological response units. During the model calibrations (when allowed) parameter values reflecting root depths, soil water retention capacity, soil hydraulic conductivities, and time constants in subsurface flow routing were adjusted.

The MIKE SHE also distributed the rainfall information to different inputs in six meteorological zones. Information on topography, soil, vegetation, and geology were distributed to a 1-km grid. Thus MIKE SHE carried out calculations at 1090 horizontal grid points. During the model calibrations (when allowed) parameter values reflecting soil depth and maximum root depths, as well as an empirical drainage time constant, were adjusted. In order to minimize the calibration work the parameter values were not varied within all 1090 grid points, but kept identical within each of the 13 land-use classes. In general, the parameters for which field data were available, such as soil water retention curves and leaf area index, were not modified during the calibration process.

The present study has aimed at testing various types of general modeling systems. However, it should be emphasized that validation results are not solely dependent on the modeling system but, indeed, also depend on the hydrologist operating the model, including his or her personal interpretation of available information and subjective assessments. In the present study this element of uncertainty has been minimized to the extent possible by assigning three experienced hydrologists with comprehensive experience in the application of each of the three modeling systems and by providing each of them with the same catchment data.

The calibration procedure adopted was that of "trial and error," implying that the hydrologists made subjective adjustments of parameter values in between the calibration runs. The numerical and graphical performance criteria described above were used as important guidance for the hydrologists when deciding upon the set of parameter values which they assessed to be the optimal ones. As these decisions inevitably depend on the personal experiences and judgments of the hydrologists, it may be argued that this procedure adds an undesirable degree of subjectivity to the results. However, given the large number of performance criteria and the large number of adjustable parameters, especially in the WATBAL and MIKE SHE models, suitable and well-proven automatic parameter optimization techniques did not exist. Instead, by applying the standard calibration procedure by which the three hydrologists had comprehensive experience, the results may be seen as typical results from three different modeling systems, when using standard engineering procedures for data collection, model construction, and calibration.

Results of Model Validation Test Scheme

The results of the six tests outlined in Figure 6 are summarized in Figure 7, which shows the overall water balances and

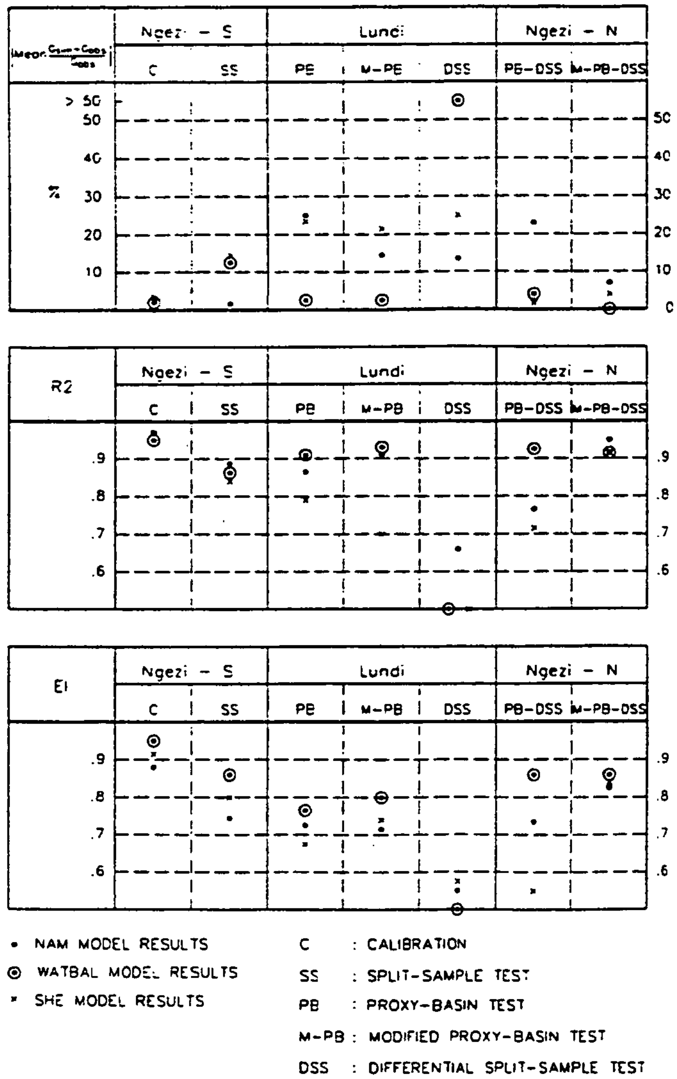

Figure 7. Summary of key validation results for all tests.

the $R2$ and EI numerical criteria. Simulated and observed hydrographs are shown in Figure 8 for two of the tests from the Lundi and Ngezi-North catchments. Annual water balances are shown for all the tests in Figures 9–15. Assessments of uncertainties in the PB predictions are shown in Figures 16 and 17. Note that the different performance criteria presented in the figures focus on different aspects, such as overall annual water balances (Figures 9–17), monthly flows ($R2$ in Figure 7), flow pattern on a daily basis (EI in Figure 7) and hydrograph shapes (Figure 8). The results are discussed test by test in the following sections.

SS Test

This test is based on data from Ngezi-South and comprises an initial calibration of the models and a subsequent validation using data for an independent period. As indicated in Figures 7, 9, and 10 the performances of the three models are very similar. All models are able to provide a close fit to the recorded flows for the calibration period, while for the independent validation period the performance is somewhat reduced, as expected. The reduction is, however, limited, and all models are able to maintain a very good representation of the overall water balance and the interannual and seasonal variations, as well as the general flow pattern.

PB Test

This test comprises a transfer of models to the Lundi catchment, adjustment of parameters to reflect the prevailing catchment characteristics, and validation without any calibration. The PB test was arranged to test the capability of the different models to represent runoff from an ungauged catchment area, and hence no calibration was allowed prior to the simulation. All models have used the experience from the Ngezi-South calibrations in combination with the available information on the particular catchment characteristics for Lundi. While the NAM model has used this information in a purely subjective manner to revise model parameters, both the WATBAL and MIKE SHE models have directly used this information for the model setup. The estimates prepared by the latter two models have, however, also been influenced by the individual modelers' subjective interpretation of the available information on soil and vegetation characteristics.

In order to assess the effects of the uncertainty in parameter estimation as perceived by the individual modelers, three alternative runoff simulations were prepared, reflecting expected low, central, and high (runoff) estimates, respectively. The results of the central estimates are included in Figures 7, 8a, and 11, while annual runoff figures for the assessed uncertainty intervals are shown in Figure 16.

In general, all models provide an excellent representation of the general flow pattern and the overall water balance, while maintaining the significant interannual variability to a satisfactory degree. The predicted hydrographs for the rainy season of 1973/1974, shown in Figure 8a, confirm that the overall hydrograph pattern is predicted quite well by all three models.

The overall performance of the central estimates by the NAM and MIKE SHE models is somewhat reduced compared to validation runs for the Ngezi-South catchment as expected when no calibration is possible. The estimates would, however, still be very valuable for all practical purposes. For the WATBAL model, the central estimate is even better than obtained for the validation period for Ngezi-South, providing for a very accurate representation of observed runoff record.

From Figure 16 it appears that the assessed uncertainty interval for the NAM predictions of annual runoff is about twice as wide as for the WATBAL and MIKE SHE predictions.

M-PB Test

This test is based on the same data from Lundi as the above PB test. The M-PB test was undertaken to evaluate whether better model performance could be obtained should short-term measurements be available for calibration. Hence, before the results of the previous test were revealed, 1 year (1975/1976) of runoff record was released for calibration, and the PB test repeated. The main results of this test are summarized in Figure 7, and annual water balances are shown in Figure 12.

For the NAM model the short-term calibration leads to an improved performance, decreasing the deviation of the overall water balance to some 15%. At the same time, the statistics of $R2$ and EI confirm the good representation of monthly flows and the overall flow pattern in general.

For the WATBAL model the short-term calibration introduces only a slight improvement in the overall performance. The reason for this is thought to be due to the originally very good performance, which in any case would be difficult to improve. The main benefit of the short runoff record is in this case primarily to confirm the validity of the central estimate

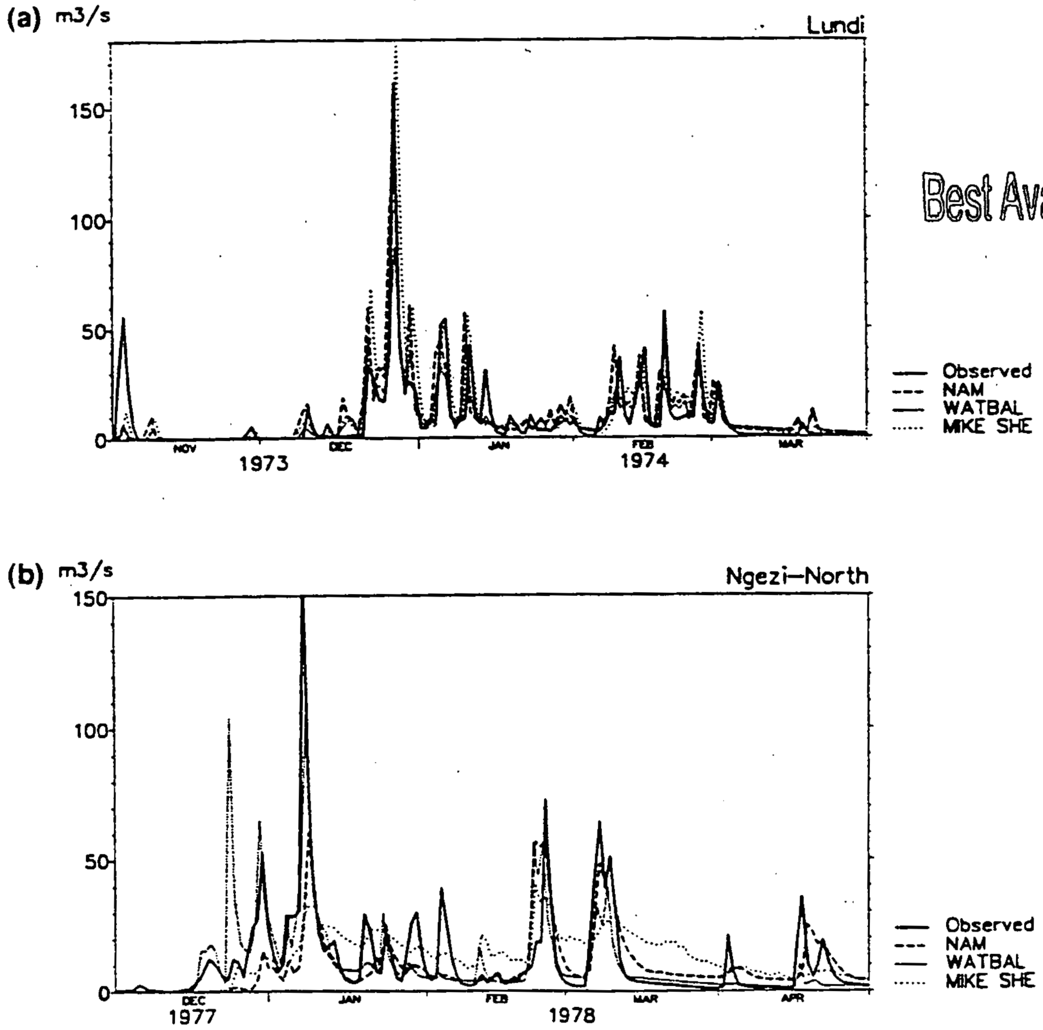

Figure 8. (a) Lundi (central estimates) proxy-basin (PB) test hydrographs from 1973/1974. (b) Ngezi-North (central estimates) PB differential split-sample (SS) test hydrographs for 1977/1978.

and hence to reduce the uncertainty related to the final runoff estimate. In this sense the calibration has proven quite valuable and would indeed be so in any practical case.

For the MIKE SHE model the calibration has not introduced any improvement in the overall performance. As compared to the best of the original estimates (i.e., the low case) the calibration has in fact caused a deterioration of the performance. This rather unfortunate incident may occur for all

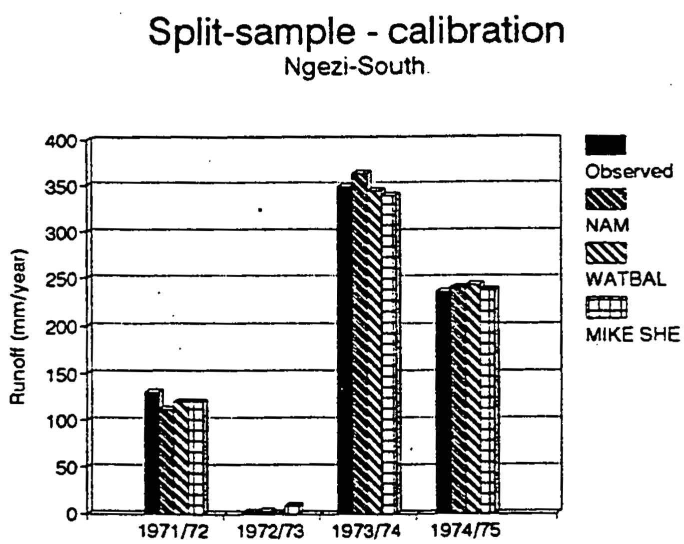

Figure 9. Annual water balances for the calibration part of the SS test on Ngezi-South catchment.

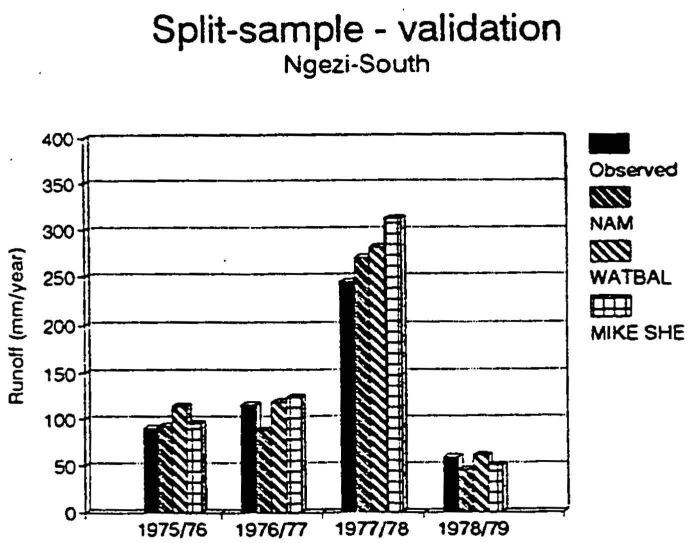

Figure 10. Annual water balances for the validation part of the SS test on Ngezi-South catchment.

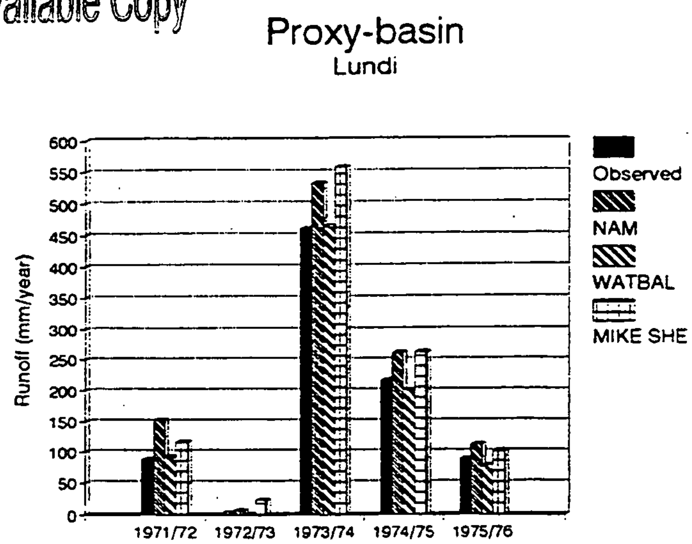

Figure 11. Annual water balances for PB test on Lundi catchment.

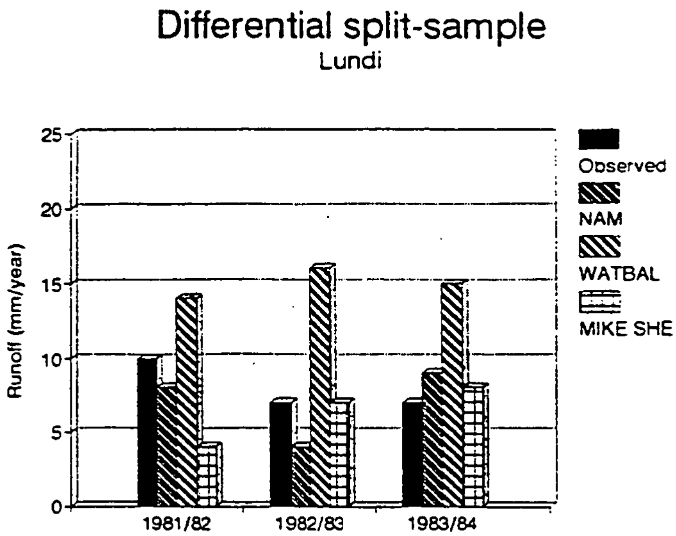

Figure 13. Annual water balances for differential split sample (DSS) test on Lundi catchment.

types of models when calibration data are not fully consistent, but it appears that the SHE type of model requires a greater reliability of input data than other, more simple types of models to avoid the pitfall of miscalibration.

DSS Test

This test consists of model calibrations based on data from Lundi for 4 wet years (1971/1972–1975/1976 with mean annual runoff of 171 mm) and validation on data from 3 very dry years (1981/1982–1983/1984 with mean annual runoff of 8 mm). The purpose of this test is to assess the capability of the models to do simulations under nonstationary climate conditions. A summary of the main results of the differential SS tests is given in Figure 7, and the annual water balances are shown in Figure 13.

As is evident from the results, both NAM and MIKE SHE predict the water balance well. The WATBAL model, however, grossly overestimates the peaks in the relative sense, causing the simulated average runoff to be about twice that measured (15 mm compared to 8 mm). The related statistics are poorer than those in the other testing schemes, but it should be noted that even small deviations cause poor statistics when mean flows are as low as those in this case.

PB-DSS Test

This test is based on data from the third catchment, Ngezi-North. Without allowing for any prior calibration, all modelers were requested to prepare low, central, and high estimates of the expected series of flows for the 1977/1978–1983/1984 period. This period contained a sequence of mainly wet years (1977/1978–1980/1981) followed by 3 consecutive dry years, with rainfalls being less than half of that experienced in the former period.

At the stage when the measured flow record was revealed, it was unfortunately discovered that the record for the 1979/1980–1980/1981 years was erroneous and hence had to be disregarded when computing the test statistics. The results of this test are summarized in Figure 7, while the annual water

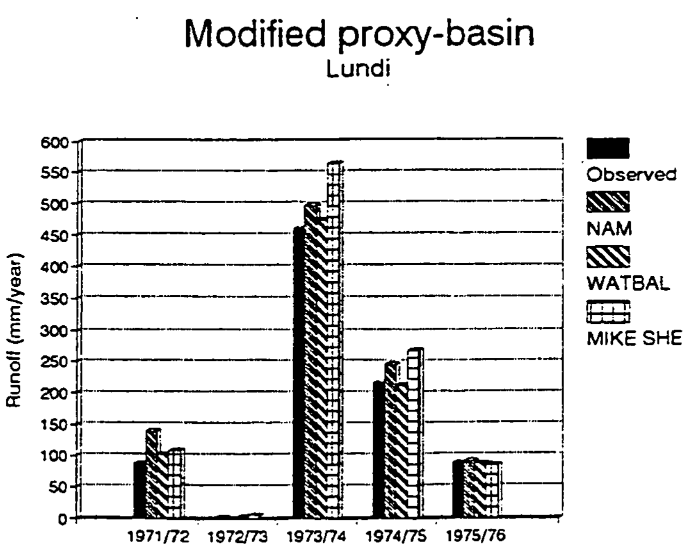

Figure 12. Annual water balances for modified proxy-basin (M-PB) test on Lundi catchment.

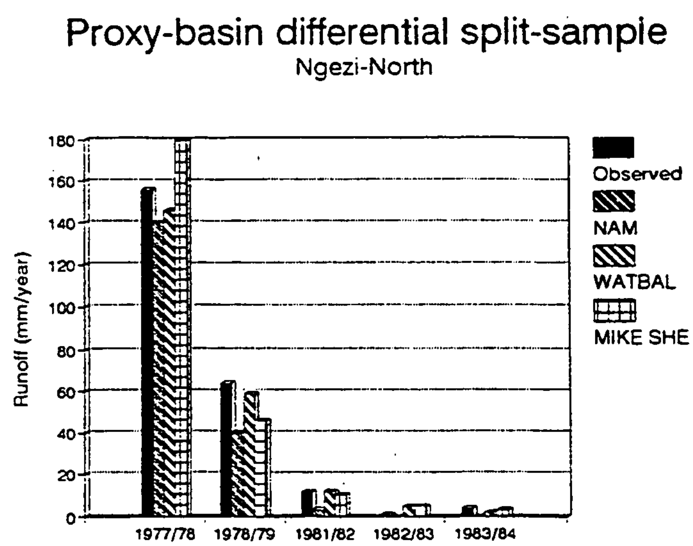

Figure 14. Annual water balances for proxy-basin differential split-sample (PB-DSS) test on Ngezi-North catchment.

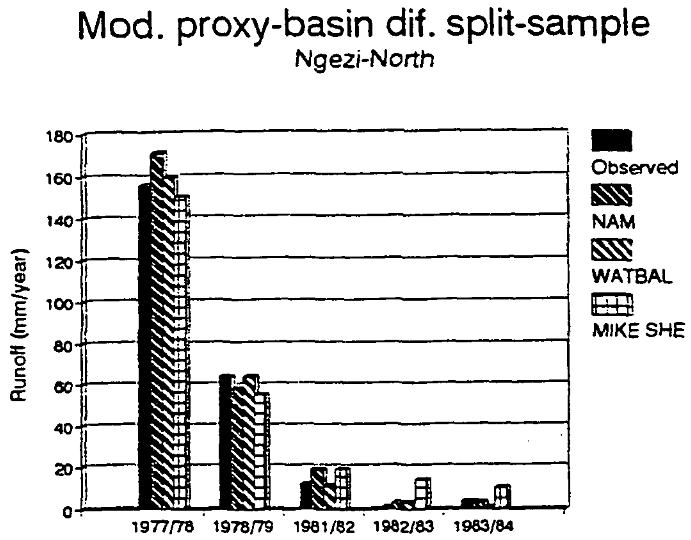

Figure 15. Annual water balances for modified proxy-basin differential split-sample (M-PB-DSS) test on Ngezi-North catchment.

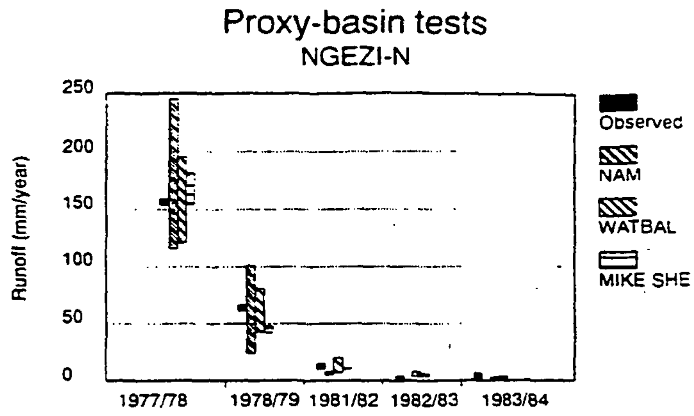

Figure 17. Assessments of uncertainty interval for prediction of annual water balances in the PB-DSS test on Ngezi-North catchment.

balances are shown in Figure 14. The assessed uncertainty intervals of the model predicted annual runoff are shown in Figure 17.

From Figure 17 it appears that all models have managed to provide for a nonbiased range of estimates of the overall water balance, which for some models is quite narrow: NAM, ±50%; WATBAL, ±30%; and MIKE SHE, ±10%. In terms of the overall water balance, the central estimates of the models agree within 25% (NAM), 5% (WATBAL), and 2% (MIKE SHE). The agreement between the recorded and simulated monthly flows and the flow duration curves, however, is less accurate for NAM and MIKE SHE than for the WATBAL model, which provides for an excellent fit in terms of these measures. The reason for the somewhat lower $R2$ and EI figures for the NAM model is related to its generally less accurate prediction of flows, while for the MIKE SHE model this is directly linked to the erroneous assessment of a key drainage parameter, causing the model to produce much more base flow than actually exist.

Hydrographs showing measured discharge and predictions by the three models for the rainy season of 1977/1978 are presented in Figure 8b. These graphs confirm the conclusions derived from the numerical criteria, $R2$, and EI, namely, that the WATBAL reproduces the observed hydrograph very well, while the daily hydrograph for MIKE SHE reveals major errors in overall flow pattern. Note that the model which produces the best overall water balance (MIKE SHE) has at the same time the poorest fit when compared on daily values.

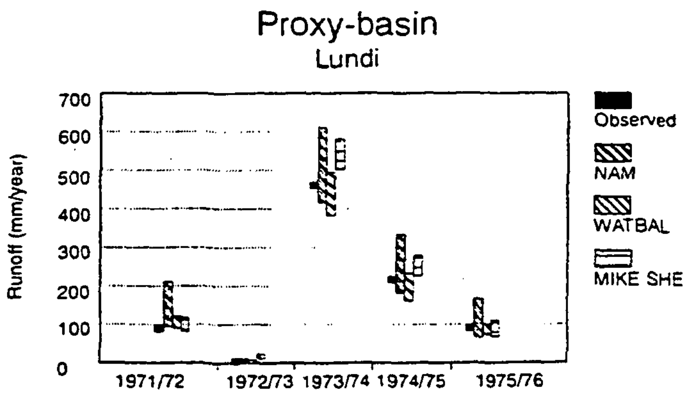

Figure 16. Assessments of uncertainty interval for prediction of annual water balances in the PB test on Lundi catchment.

M-PB-DSS Test

This test is based on the same data from Ngezi-North as the previous PB-DSS test. Following the calibration of all models based on only 1 year of data (1977/1978), before the results for other years were revealed the above test was repeated. The main results of the modified test are shown in Figures 7 and 15. These results clearly demonstrate that access to only 1 year of runoff data has enabled all models to provide an excellent representation of the runoff within the entire testing period.

The overall water balance agrees within 7% for all models and despite the fact that the calibration was based on a wet year, annual flows for the dry period come within the right order of magnitude, although the relative deviation in some cases is quite significant. The high $R2$ and EI scores achieved by all models confirm that the representation of the monthly flow sequence and the overall flow pattern has become very good after the calibration.

Discussion and Conclusions

The three generalized modeling systems, NAM, WATBAL, and MIKE SHE, have been subject to a rigorous testing scheme on data from three Zimbabwean catchments. NAM is a typical representative for the lumped conceptual class of models, while MIKE SHE similarly belongs to the distributed physically based class. WATBAL falls between the two classes. However, for the specific applications in Zimbabwe, where surface water hydrological aspects have been dominated, it can be argued that WATBAL can be considered as another representative of the distributed physically based class.

Although establishing an objective framework for the model tests and intercomparisons has been attempted, it should be recognized that the results of a certain validation will be influenced by the specific test conditions, including the particular climate, catchment characteristics, data availability, and quality as well as subjective assessments made by the user (e.g., interpretation of available information for determining model parameters). Hence the obtained results are not only a function

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of the modeling system itself, but also of the user and numerous other factors. To arrive at a firm conclusion many validations would usually be required, and the limited number of tests undertaken therefore suggests that individual results may only be cautiously concluded.

With this caution regarding generality in mind, a number of specific conclusions may be derived from the case study. First, in view of the difficult tasks given to the models involving simulation for ungauged catchments and nonstationary time periods, the overall performance of the models is considered quite impressive. The overall water balance agrees within $\pm 25\%$ in all cases but one, and good results are achieved without balancing out excessive positive and negative deviations within individual years. In most cases the models score an $R2$ value at about 0.8 or greater and an EI index generally above 0.7.

Secondly, the following is noted with regard to the specific types of validations tests:

1. For the SS test the NAM, WATBAL, and MIKE SHE systems generally exhibit similar performance. All models are able to provide a close fit to the recorded flows for the calibration period, without severely reducing the performance during the independent validation period. Hence this test suggests that if an adequate runoff period for a few (3–5) years exists, any of the modeling systems could be used as a reliable tool for filling in gaps in such records or used to extend runoff series based on long-term rainfall series. Considering the data requirements and efforts involved in the setup of the different models, however, a simple model of the NAM type should generally be selected for such tasks.

2. For the PB tests, designed for validating the capability of the models to represent flow series of ungauged catchments, it had been expected that the physically based models would produce better results than the simple type of models. The results, however, do not provide unambiguous support for this hypothesis. All three modeling systems generated good results, with the WATBAL providing slightly more accurate results than the others. Hence for the Zimbabwean conditions the additional capabilities of the MIKE SHE, as compared to the WATBAL, namely, the distributed physically based features relating to subsurface flow, proved to be of little value in simulating the water balance. For the PB tests it is noticed that the uncertainty range represented by the low and high estimates is significantly larger for the NAM than for the WATBAL and MIKE SHE cases. This probably reflects the fact that parameter estimation for ungauged catchments is generally more uncertain for the NAM, whose parameters are semiempirical coefficients without direct links to catchment characteristics.

3. A general experience of the M-PB tests is that allowing for model calibration based on only 1 year of runoff data improves the overall performance of all models. The improvement appears to be particularly significant for the NAM model, which also showed the largest uncertainties in the cases where no calibration was possible.

4. For the DSS tests all models have been able to simulate flows of the right order of magnitude and correct pattern. Hence all models have proven their ability to simulate the runoff pattern in periods with much reduced rainfall and runoff as compared to the calibration period. On the basis of these results there appears no immediate justification for using an advanced type of model to represent flows following a significant change of rainfall, providing a number of years are available for calibration purposes. It is tempting to extend this finding to suggest that the simple type of model could be used to assess the impact of climate change on water resources. It should be recognized, however, that above results cannot fully justify such a hypothesis, since a long-term climate change would probably bring about changes in vegetation and their evaporation. This type of nonstationarity has not been adequately tested.

As far as the SS tests are concerned the above conclusion is in full agreement with results of other studies [e.g., *Michaud and Sorooshian*, 1994]. With regard to the PB tests the present conclusion in favor of the distributed physically based modeling systems is in agreement with, albeit more vague than, that of *Michaud and Sorooshian* [1994].

In summary, the present study, as well as similar studies reported in literature, suggests the following conclusions with regard to rainfall runoff modeling.

1. Given a few (1–3) years of runoff measurements, a lumped model of the NAM type would be a suitable tool from the point of view of technical and economical feasibility. This applies for catchments with homogeneous climatic input as well as cases where significant variations in the exogenous input is encountered.

2. For ungauged catchments, however, where accurate simulations are critical for water resources decisions, a distributed model is expected to give better results than a lumped model if appropriate information on catchment characteristics can be obtained.

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THE USE OF MATHEMATICAL MODELS IN FLOOD ACTION PLANNING AND FLOOD FORECASTING

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After the vast floodings in Bangladesh in 1987 and 1988, a comprehensive flood action plan was launched with support from the international donor community under coordination of UNDP. The action plan consisted of in total 27 different projects, all with focus on flood forecasting, flood control and flood management. Most of the projects used mathematical models established by the Surface Water Modelling Centre in Bangladesh for the analyses of shortterm and longterm impacts. These models allow for local detailed analyses as well as combined regional analyses of all aspects related to surface water management. The modelling package has been extended with an advanced real time update routine, allowing its use for real time flood forecasting. An operational flood forecasting system has been established for the entire major rivers system in Bangladesh. As one of the components under the Flood Action Plan, a socalled Flood Management Model has been established. Through coupling with GIS and other data management and analyses facilities, these models are tailored for management use in connection with water resources planning and flood emergency operations.

1. INTRODUCTION

Owing to its geographical location, Bangladesh is exposed to a wide range of extreme natural phenomena; it is located on a fragile portion of land in the worlds largest delta, which comprises three of the worlds most unstable rivers. The rivers flowing into Bangladesh drain some of the wettest catchments on earth with average yearly rainfalls as high as 11 m. In addition, Bangladesh is one of the most fertile regions in the world.

Controlling of water is vital to Bangladesh. The sharing of water with the neighbour countries, India, Nepal, Bhutan and China is subject to frequent disputes. Within the country, numerous schemes for flood protection, drainage and irrigation are implemented every year. Despite these efforts, it is neither possible nor desirable to entirely prevent flooding in Bangladesh.

This view is supported by the findings of the recent comprehensive Flood Action Plan, which recommends a combination of structural and non-structural approaches to the water management. The planning and design of such requires a delicate and difficult balance. It requires both an overall and a detailed understanding of the complicated physical and sociological processes which are interlinked by the water. These interrelationships make many of the problems too complex for the traditional project-to-project type of planning. Management tools, capable of integrating all relevant processes and connecting easily with other important disciplines, are needed.

An important element in this has been the development of multipurpose mathematical models for the entire country of Bangladesh with one general model for the major river system, and six regional models for the separate regions, see Fig. 2. These models are increasingly used in the water planning and management, and have become an important element in the design, monitoring and real time control.

The Surface Water Modelling Centre (SWMC) was established in 1990 to institutionalize the use of these models.

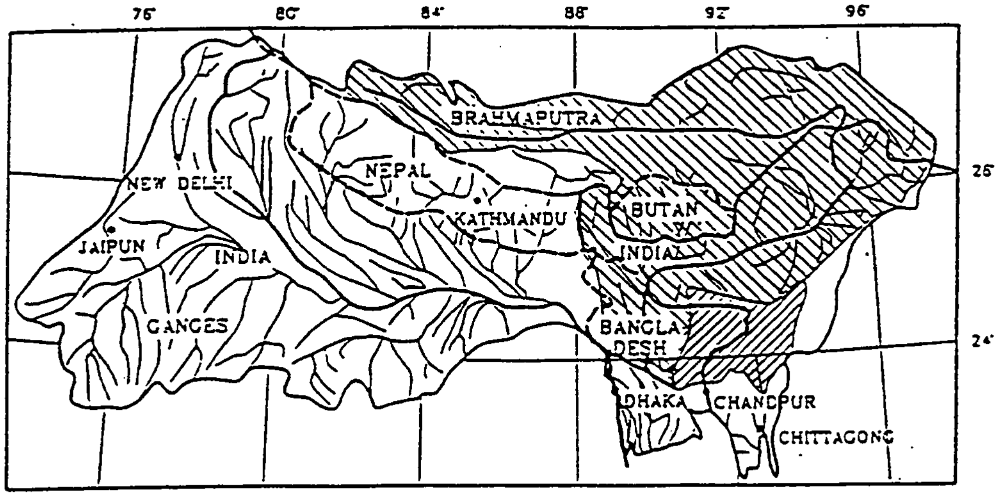

Fig. 1 The Ganges, Brahmaputra and Meghna river basins.

Fig. 2 Location of the Six Regional Models in Bangladesh.

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In connection with establishment of the models, comprehensive survey programmes have been carried out for the entire river system in the country, comprising surveys of river cross sections, hydrographic data, sediment data, establishment of benchmark connections and at locations also water quality and salinity measurements.

The first versions of the models were established just in time for use in the comprehensive Flood Action Plan for Bangladesh, which was launched after the 1987 and 1988 floods. Out of the 26 different components of the Action Plan, the models were used in 15. It was possible to use the models independently by the studies or with support from the Centre, as appropriate. A condition for the use of the models was that additional data, which were measured and used to update and refine them, were transferred back to the Centre. In this way the Centre maintains a currently updated database and model suite for use in future water resources and flood studies.

Two of the 26 studies under the Flood Action Plan, which relied especially on modelling, will be further described in the following.

2. FLOOD FORECASTING AND MANAGEMENT

Flood forecasting is vital for Bangladesh. In nearly all monsoon seasons, 30 to 50 per cent of the country is inundated causing severe damage and making thousands of people homeless.
The floods are caused by three main types of events:

Flash Floods

These occur mainly in the eastern and northern rivers where short duration heavy rainfalls in the mountain catchments (within India), lead to rapidly rising hydrographs, rapid runoff response and very fast flood waves with consequential damage.

Rainfall Floods

These are caused locally by high rainfall intensities and long duration monsoon rainfall causing flooding due to inadequate local drainage.

Monsoon Floods

These are caused by overflows from the major rivers and their tributaries causing often extensive areas to inundate. The rivers rise slowly and may stay at high flows for extended periods of many weeks. Simultaneous peaks on the three main rivers can cause particularly extensive flooding.

Storm surges

These are caused by cyclones in April-May and October-November, which generate tidal surges, which inundate low lying areas at the coast and along the rivers in the coastal region.

A Disaster Management Bureau, DMB, was established in 1992 to be the focal point for the government's disaster management activities and with a wide brief:

- Enhancing the capacity of government and local-level authorities to warn people of imminent threats of cyclones and floods
- Ensuring the effective dissemination of appropriate warnings of floods and cyclones
- Activating and operating a national Emergencies Operations Centre
- Developing planning and preparedness activities at all levels of the community

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The Disaster Management Bureau relies on operational forecasts made by the Flood Forecasting and Warning Centre (FF&WC), which was established under the Bangladesh Water Development Board in 1972.

One of the Flood Action Plan components: FAP 10, aims at reinforcing and expanding the flood forecasting and warning services of this Centre. The forecasting system is based on the MIKE 11 FF Flood Forecasting version, which has been developed especially for this purpose. The core of the system is the General Model for the main river system in Bangladesh (described above).

During monsoon, the system is operated daily. Every morning, information about rainfall and water levels from more than 40 stations in the country are transmitted through wireless radio to the FF&WC. In addition, weather forecasts and satellite images are interpreted to derive the best estimate of the rainfall for the days ahead. The rainfall data and boundary data at the Bay of Bengal and at the geographical boundaries of the country are used as driving input for the simulation model. The water level recordings within the country are used for "real time update" to improve the models performance and accuracy. Daily forecasts of water levels are issued 24, 48 and 72 hours ahead.

Thanks to the updating facility and to MIKE 11's ability to accurately describe flood plain flows, the quality of the forecasts have been very reliable. A general comparison for 1991, which was one of the early years of applying the full system, yielded the following mean deviations:

- 24 hour forecast 4.7 cm
- 48 hour forecast 9.4 cm
- 72 hour forecast 14.1 cm

At present the flood forecasting system is being extended to cover also some of the large tributaries to the main river system, see Fig. 3.

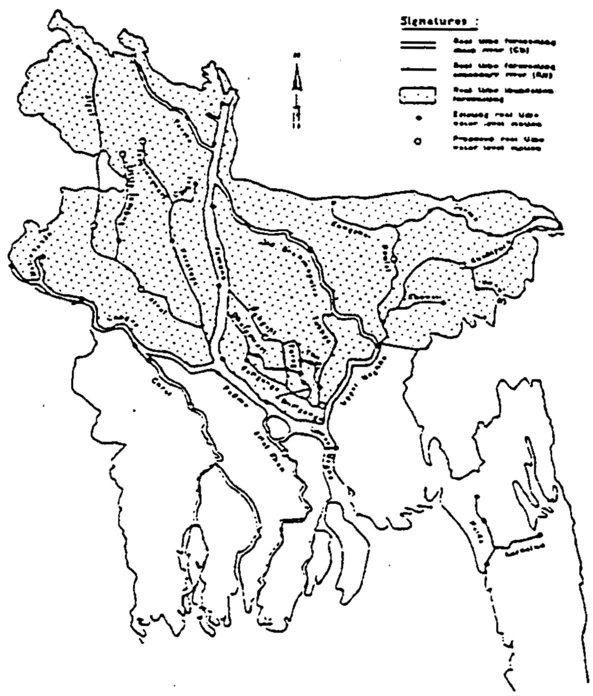

Fig. 3 Refined Flood Forecasting Model area with indication of the area where inundation forecasting will be undertaken

This allows more detailed forecasts especially in the northern regions. At the same time, the flood plain description in the model will be refined and coupled with GIS, allowing inundation forecasting. The forecast of inundation will be on "Thana" scale in form of flood maps (see Fig. 4).

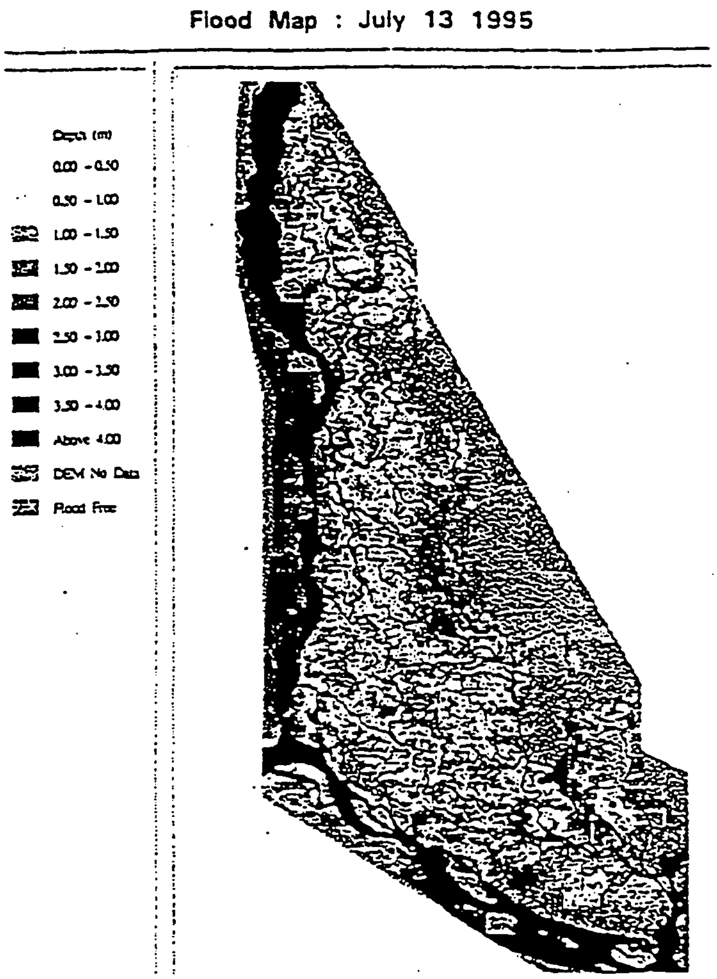

Fig. 4 Example of flood inundation map produced with GIS coupling.

3. FLOOD MANAGEMENT MODELLING

One of the central elements in the Flood Action Plan is the so-called "compartmentalisation". The idea is to construct series of ring dikes with gate structures, allowing for a controlled flooding and drainage within and between compartments and between the compartments and the main rivers and drainage channels. The flows could be through non-gated, throttling structures and gated structures on the rim of the subcompartments and compartments. The identification of overall water-management strategies for compartments and the development of simple operational rules are essential for the successful implementation of compartmentalisation on a large scale.

To address this need for detailed information on flood plain inundations and the development of operational guidelines for flood control structures and schemes and for compartmentalisation, the Flood Management Model (FMM) was developed. In the most basic terms, the FMM is a user-friendly, graphics-based tool designed to assist decision makers in the management of floods. The target group of FMM includes:

- flood management decision makers
- operators of FCD schemes in general and of compartments in particular
- flood forecasting authorities
- planners and designers of FCD schemes and other infrastructure developments (roads, highways, railways), which may have an impact on flooding and drainage conditions
- inland water transport authorities
- universities (research, education and training)

In the planning and management for flood-prone areas, a highly time consuming task is to examine the flood impact on the numerous human functions in the area: infrastructure, housing, crop, water intakes, electrical installations, emergency shelters etc. etc. An effective and convenient way to provide an overview is by use of the database facilities in GIS. This allows overlay of different layers of information, which is stored in a graphical form. One of these layers comprises the flood inundation pattern (water levels and extend of flooding) computed with the hydrodynamic models. By subtracting another layer of data with terrain elevations, the local depth of flooding can also be derived. The system design allows for a rapid generation of inundation boundaries showing different flood scenarios, such as scenarios with or without flood protection measures, and thus guarantees a consistent and effective approach to locating inundated land as compared to manual methods.

The flood impact maps are then overlaid with other spatial information on land-use, housing, infrastructure etc. In this way, damage assessments can be made, and by coupling economic information, the costs and benefits can be directly quantified and compared. For the derivation of e.g. crop damage assessments, flood depth maps and flood duration maps are used in combination. By overlaying these with crop maps, it is estimated where the crop can withstand the flood without being damaged (the limit is typically three days). This information is translated into economical figures for the overall evaluation and optimisation. An example of a flood damage map is given in Fig. 5.

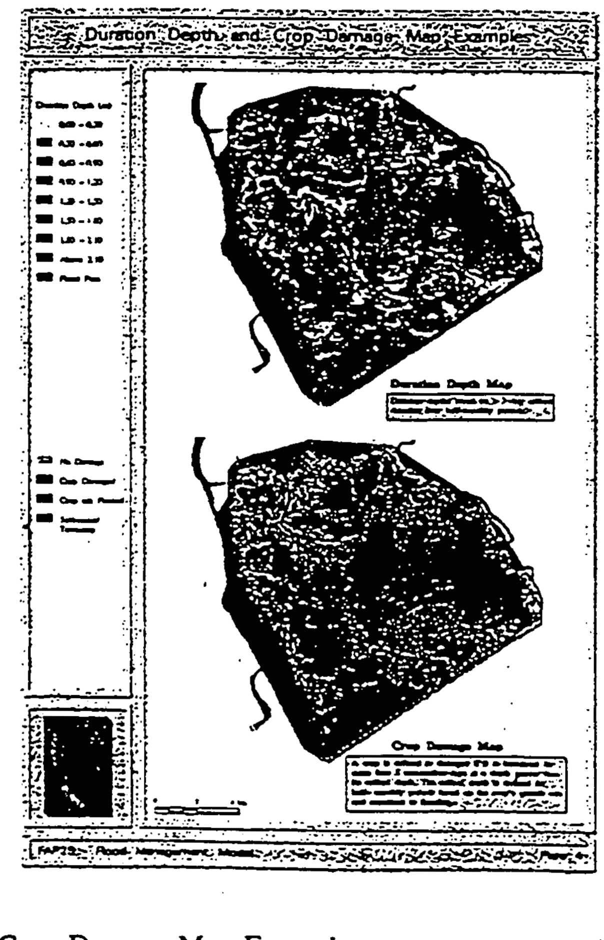

Fig. 5 Duration Depth and Crop Damage Map Examples.

The FMM can be used for multiple purposes:

- designation of low-impact and high-impact development areas
- the estimation of cost-benefit ratios for flood mitigation proposals
- the overall optimisation of development plans
- the planning of emergency operations
- the on-line monitoring and real-time control.

In connection with the optimal planning and design, it is crucial to establish a set of design criteria, which reflect the chosen statistical return period. These design criteria are again spatially distributed and best represented in the GIS environment. By coupling the statistical (Extreme Value EVA) evaluation directly to the GIS, it is possible to derive flood maps for any statistical return period, which account for all local joint probability effects. Such methods replace the classical "design flood" methods, which were especially poor in complex areas with large topographical variability and/or with more than one source of impact (e.g. both flood and tide). The methodology of this approach is sketched in Fig. 6.

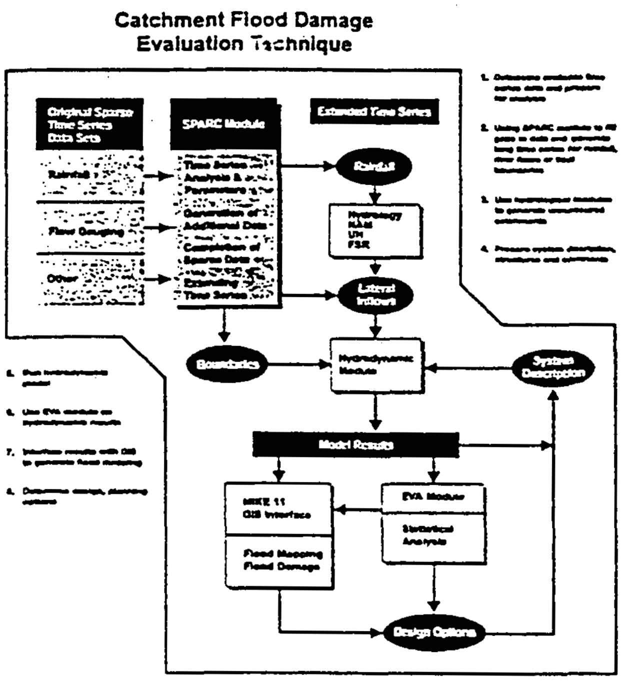

Fig. 6 Outline of Flood Damage Evaluation Technique.

4. MORPHOLOGICAL MODELLING AND FORECASTING

The large rivers in Bangladesh are morphologically very unstable, and frequently change coarse. Bank erosion is a severe problem at many locations, not least along the Brahmaputra river. The problem is accentuated in connection with the Jamuna Bridge crossing presently under construction. The concern on longer term is whether the river will maintain a stable course in the vicinity of the bridge. In order to ensure this, huge guide walls are being constructed starting about 1.5 km upstream of the bridge.

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The concern on shorter term is the morphological stability of the river during construction. Since the construction was planned, the river has already changed coarse considerably, and a deep channel has been scoured at the location where one of the two guide walls was planned to be constructed. This has required a revision of the layout and the construction plans, and a forecast tool is required to ensure adequate warning time, if further revisions are to be made.

The morphological forecast tool is based on an overall approach, which combines the use of one- and two-dimensional mathematical hydrodynamic and morphological models with analyses of satellite imageries to determine bank line changes and associated bank erosion rates. Large physical model tests are also being carried out (at the River Research Institute), which support the mathematical modelling.

The central modelling tool is MIKE 21-Curvilinear, a general purpose suite of modules, which describes the interaction between hydrodynamics and morphology. By aid of its curvilinear grid, it is able to provide good resolution of the local flow pattern close to the bank, and with a (quasi 3-d) description of helical flows near the banks, it is also in the state to simulate bank erosion.

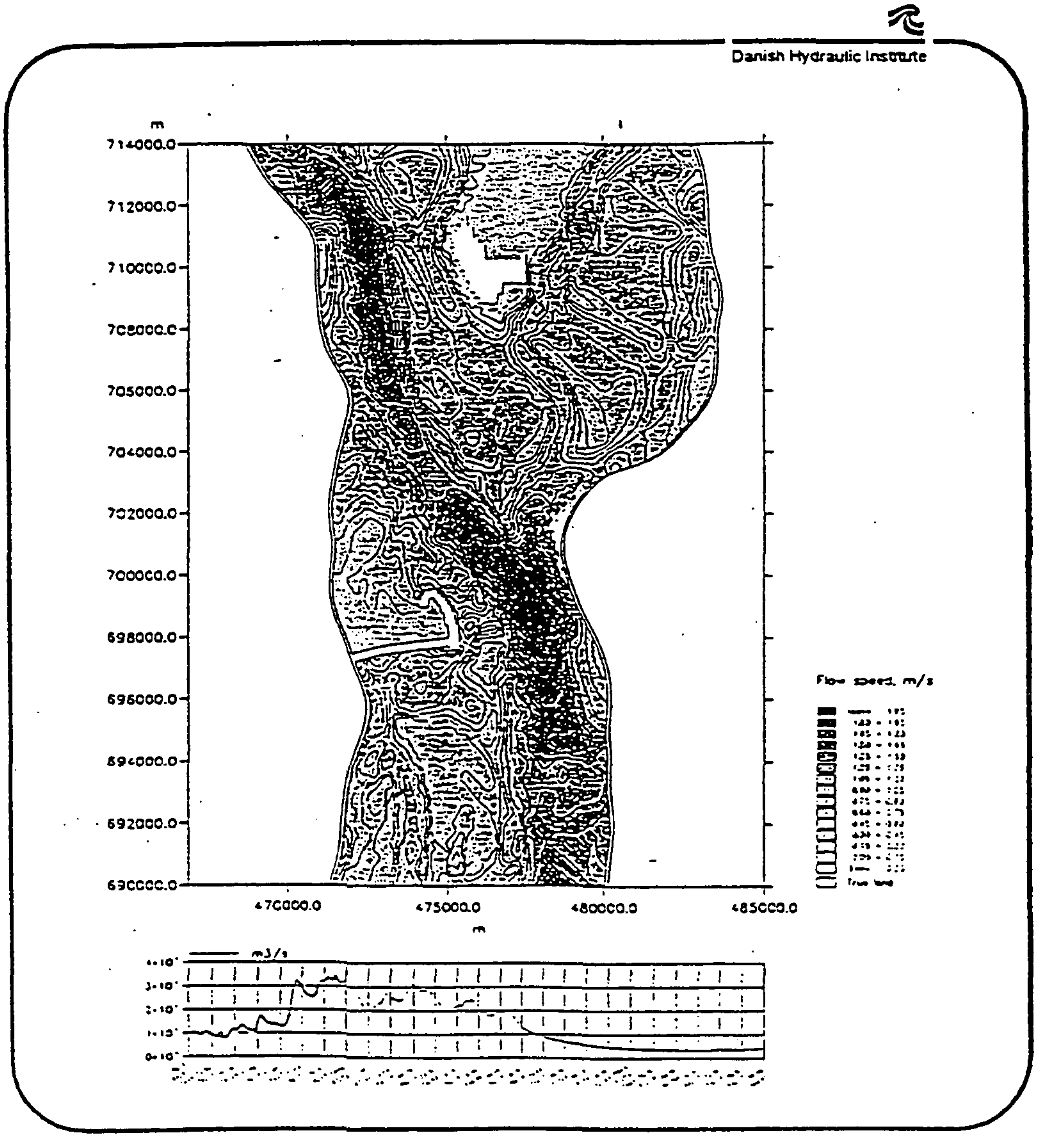

Fig. 7 Simulated change in flow speed due to bridge construction.

The curvilinear model has been calibrated on the basis of detailed sediment transport and morphological measurements in the area. It has been used to forecast hydrodynamic and morphological conditions in the monsoon of 1996. Fig. 7 shows the forecasted change in the river morphology in 1996 due to the construction of river training works for the Jamuna Bridge.

5. CONCLUSIONS

The water management in Bangladesh, its planning and design requires a delicate and difficult balance. It requires both an overall and a detailed understanding of the complicated physical and sociological processes which are interlinked by the water. These interrelationships make many of the problems too complex for the traditional project-to-project type of planning. Management tools, capable of integrating all relevant processes and connecting easily with other important disciplines, are needed.

An important element in this has been the development of multipurpose mathematical models for the entire country of Bangladesh. These models are increasingly used in the water planning and management, and have become an important element in the planning and design. Decision support systems have been build, which merge and integrate knowledge arising from different disciplines by linking their corresponding domain knowledge encapsulators. A convenient and effective tool in linking this information and experts from different fields is GIS. A massive further development leading to a great variety of new possibilities is expected in this area in future.

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Application of GIS in hydrological and hydraulic modelling: DLIS and MIKE11-GIS

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Abstract The DLIS (Danubian Lowland Information System) and the MIKE 11-GIS represents two different applications of GIS technology in relation to hydrological and hydraulic modelling. The DLIS integrates ARC/INFO with the INFORMIX Relational Database and serves as a centralized GIS and database system with interfaces to hydrological and hydraulic models. The most important interface is between DLIS and the hydrological modelling system MIKE SHE. The MIKE 11-GIS combines GIS technology with a complex hydraulic modelling system. MIKE 11-GIS is specifically tailored for presenting and analysing model output and provides a unique tool for flood mapping and flood impact assessment.

INTRODUCTION

Mathematical models have for decades been used to study and solve environmental problems. However, during the past five to ten years there has been tremendous progress in computer performance, and during the same period the price of computers has decreased. Hence, the application of computer models in environmental engineering has become more cost effective and has gradually increased. This development has urged scientists to develop more comprehensive modelling systems in order to address more complex environmental problems. Such complex modelling studies call for a highly refined resolution of geometrical and physical data in the mathematical model. The same applies for data for model calibration and validation. Moreover, many different types of model output may be produced and often model results must be processed and presented in many different ways in order to serve different purposes. Hence, the modeller is not only facing the challenge of operating very complex models; he also has to deal with large amounts of data, with interpretation of large amounts of model output and finally, model results must be presented in a form which is also readily understandable for non-experts. For transparent visualization of model results a GIS often provides the required functionalities for maintaining, accessing, processing and presenting any type of spatial data. This paper describes two different applications of GIS within hydrological and hydraulic modelling, namely the "Danubian Lowland Information System" (DLIS) developed as part of a comprehensive environmental study in Slovakia, and the MIKE11-GIS which was developed as part of a Flood Action Plan in Bangladesh. The core mathematical modelling systems in this regard are the MIKE

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SHE and the MIKE 11 modelling systems, both developed by the Danish Hydraulic Institute. The DLIS involves to a lesser degree also the MIKE 21 modelling system.

THE MATHEMATICAL MODELLING SYSTEMS

MIKE SHE

MIKE SHE is a deterministic, fully distributed, physically based modelling system for describing the major flow processes of the entire land phase of the hydrological cycle. MIKE SHE solves the partial differential equations for the processes of overland and channel flow, unsaturated and saturated subsurface flow, and the model is completed by a description of the processes of snowmelt, interception and evapotranspiration. The flow equations are solved numerically using finite difference methods. In the horizontal plane, the catchment is discretized in a network of grid squares. River branches are assumed to run along the boundaries of the squares. Within each square the soil profile is described in a number of nodes, which above the groundwater table may become partly saturated. One-dimensional unsaturated flow calculations may be carried out for each vertical profile and each is dynamically coupled to a three-dimensional groundwater flow model. Exchange of surface and subsurface water can also take place as river-aquifer exchange. MIKE SHE provides results of the various components of the hydrological cycle. Main model outputs are typically groundwater levels and groundwater flow, actual evapotranspiration, water content in the unsaturated zone and flow and water levels in rivers. A more comprehensive description of the MIKE SHE modelling system is provided in Abbott *et al.* (1986a, 1986b) or MIKE SHE (1993).

MIKE 11 and MIKE 21

MIKE11 is a comprehensive, physically based, one-dimensional modelling system for simulation of free surface flows, sediment transport and water quality in estuaries, rivers, irrigation systems and other surface water bodies. It is a fourth generation modelling package designed for DOS and UNIX based computer platforms. The hydrodynamic module of MIKE 11 is based on the complete partial differential equations of open channel flow (Saint Venant). The MIKE 11 model operates on the basis of information on river bed and flood plain topography, including man-made hydraulic structures such as embankments, weirs, gates etc. The basic output of the MIKE 11 hydrodynamic module is water levels, discharge and flow velocities in rivers and on flood plains distributed in time and space. A variety of add-on modules are available for MIKE 11 including water quality and sediment transport modules. MIKE 21 is a hydrodynamic mathematical modelling system similar to MIKE 11, but operating in two dimensions. A comprehensive description of the MIKE 11 and the MIKE 21 modelling systems is provided in MIKE 11 (1993) and MIKE 21 (1993), respectively.

DANUBIAN LOWLAND INFORMATION SYSTEM (DLIS)

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The Danubian Lowland Information System (DLIS) is based on an integration of

ARC/INFO or ARC/VIEW and the INFORMIX relational database, linked to the modelling systems MIKE SHE, MIKE 11 and MIKE 21 of which the link to MIKE SHE is the most important. The DLIS has been developed by Dutch, Danish and Slovakian scientists as part of the project "Danubian Lowland - Ground Water Model" which is sponsored under the EU-PHARE programme. The project deals with a variety of environmental issues in the Danubian lowland related to groundwater flow and groundwater chemistry, river and reservoir hydrodynamics, water quality and sediment transport, agricultural production and nitrate leaching, and aquatic and flood plain ecology. The project area comprises about 3000 km^2 and is located between Bratislava and Komárno in the Slovak Republic. During the last few decades a huge amount of data has been collected within the project area providing a fine basis for setting up, calibrating and validating the applied mathematical modelling systems. Moreover, the output of the various established models is very comprehensive. Thus, the DLIS was developed in order to provide a centralized GIS for maintaining, processing and presentation of measured data as well as output data from the various mathematical models.

The INFORMIX component of DLIS

The INFORMIX relational database management system is used to store all data structures that can be related to a single point. In the database all data structures are attached to an Information Point (IPT). An IPT is defined by the type of data recorded and by the location of the point (x, y, (optionally z)). In DLIS 12 different groups of data have been distinguished leading to 12 different types of IPT. For instance a channel information point is a point where topographic information about the river bed and the flood plain have been recorded. At a hydrological information point, river discharge, water levels, suspended sediment concentration or a Q-h relation have been recorded and at a soil profile information point soil physical characteristics such as hydraulic conductivities or water retention curves have been recorded. Other IPTs allow storage of almost any kind of relevant data such as ground surface elevation, time series of rainfall, temperature or potential evapotranspiration, time series of groundwater quality data etc. All data stored in INFORMIX are easily accessed from the DLIS user interface.

The ARC/INFO component of DLIS

The ARC/INFO component of DLIS uses a geo-relational spatial model which supports the management of spatial and tabular data. All spatial data are stored in the ARC/INFO database while tabular data are stored in INFORMIX attached to a certain IPT. The following spatial data structures (themes) have been implemented in the DLIS:

- general data (*dls*);
- groundwater data (*grw*);
- surface water data (*sfw*);
- subsoil data (*ssl*);
- thematic data (*thm*);
- geohydrological data (*glm*);
- background data (*bck*).

The ARC/INFO part of DLIS is layer oriented enabling combination (display) of different themes stored in ARC/INFO and IPT data stored in INFORMIX. Figure 1 gives an overview of the different themes stored in DLIS and of the different possibilities of displaying themes.

Using DLIS to support mathematical modelling

The interface between the mathematical modelling systems and the DLIS is basically a number of file format conversion routines enabling transfer of data from DLIS to the models and vice versa. When using DLIS to support the mathematical modelling, three principally different application phases may be distinguished.

Model set up phase Assuming that all necessary data have been collected and implemented in the DLIS, the first phase in a model application is always to set the model up. This phase involves processing geometric and parametric data and implementing them into the mathematical model. During this model set up phase GIS functionalities are important. For a detailed MIKE SHE set up, various maps must be produced, for instance maps of surface topography, geological layer boundaries, hydrogeological properties, vegetation, coverage of meteorological stations etc. The DLIS includes different functionalities for filtering or selecting specific IPT data before processing them into maps. Many different selection methods exist but typically a selection of IPTs to be further processed involves the selection of data within a certain area and from a certain period of time. Hence, using such filters IPT data stored in INFORMIX can be processed. For instance, an average measured groundwater level for a certain month or a certain year may be produced. Any type of map produced in DLIS may subsequently be exported to a MIKE SHE file format and used in the further modelling activities. Time series stored in INFORMIX may also be stored in MIKE SHE file format.

Model calibration phase Model calibration typically involves comparison of simulation results with measured data until the model reproduces measured data satisfactorily. In order to fit the simulated data to the measured data different physical parameters in the model are changed (calibrated). In groundwater modelling, the key calibration parameters are typically the hydraulic conductivities of the saturated zone. In MIKE SHE, hydraulic conductivities are represented as maps. In order to edit such maps a specialized and fast tool is necessary. For this purpose, MIKE SHE has its own mini GIS which allows graphical editing and overlay of landmarks (digitized data). When calibrating a regional hydrological model there are typically certain areas, or sometimes just single points, where the fit between simulated and measured data is not satisfactory. This may obviously be due to insufficient model calibration but very often such discrepancies are caused by deviation between the model and nature. For instance, the presence of a canal, a groundwater abstraction or an impermeable area which is not included in the model may locally cause large differences between modelled and simulated results. In order to study measured data and hence identify and explain such deviations, the DLIS has been a useful tool. Figure 2 shows how measured data can be displayed in DLIS.

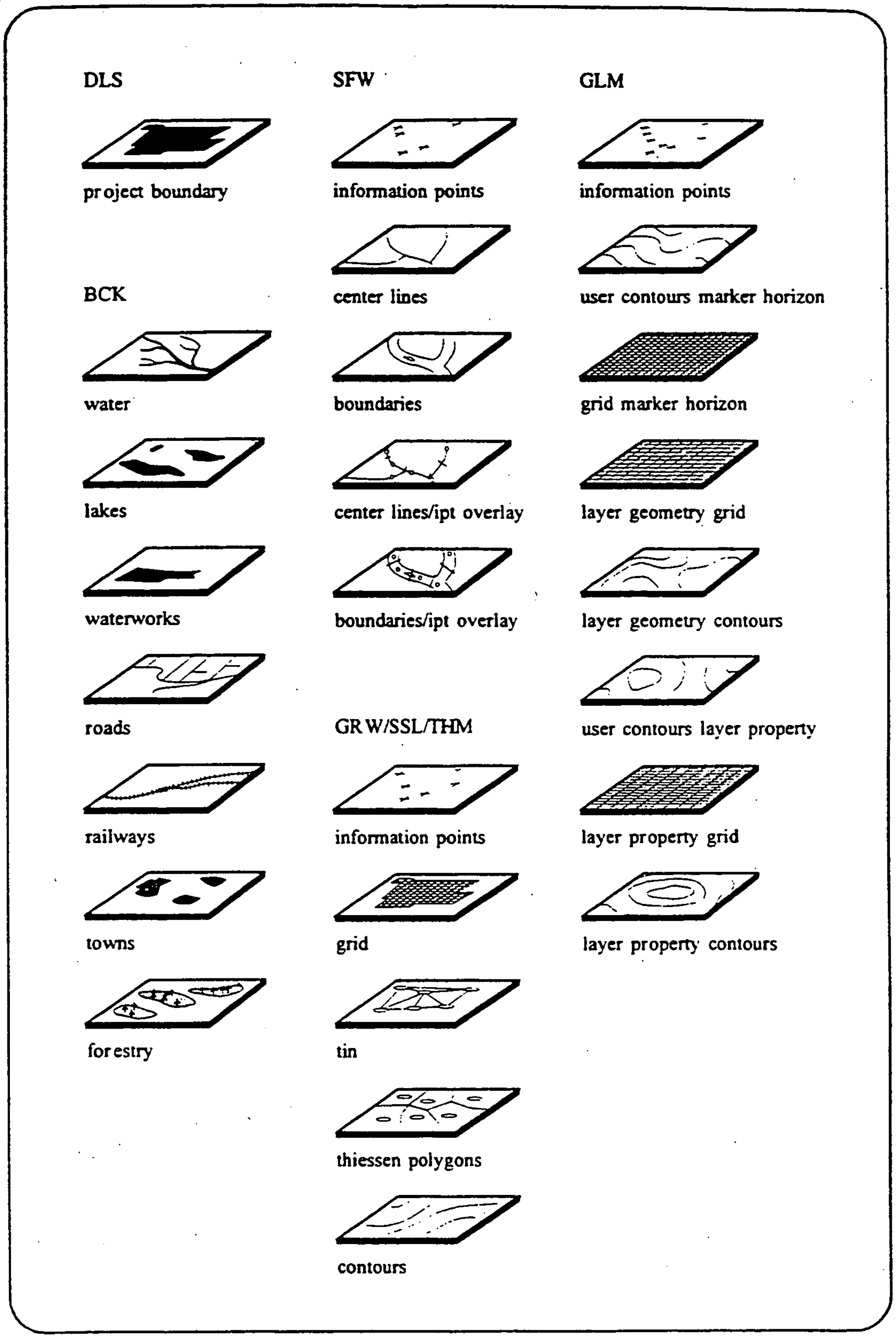

Fig. 1 Spatial data structures in DLIS.

Result presentation phase Model results must be presented in a manner which is readily understandable for non-experts as well as experts. For this purpose all general ARC/INFO presentation facilities are available through the DLIS user interface. Any kind of map which combines model results with different DLIS themes may be produced. Legends, various symbols, labels and other items may be added in order to

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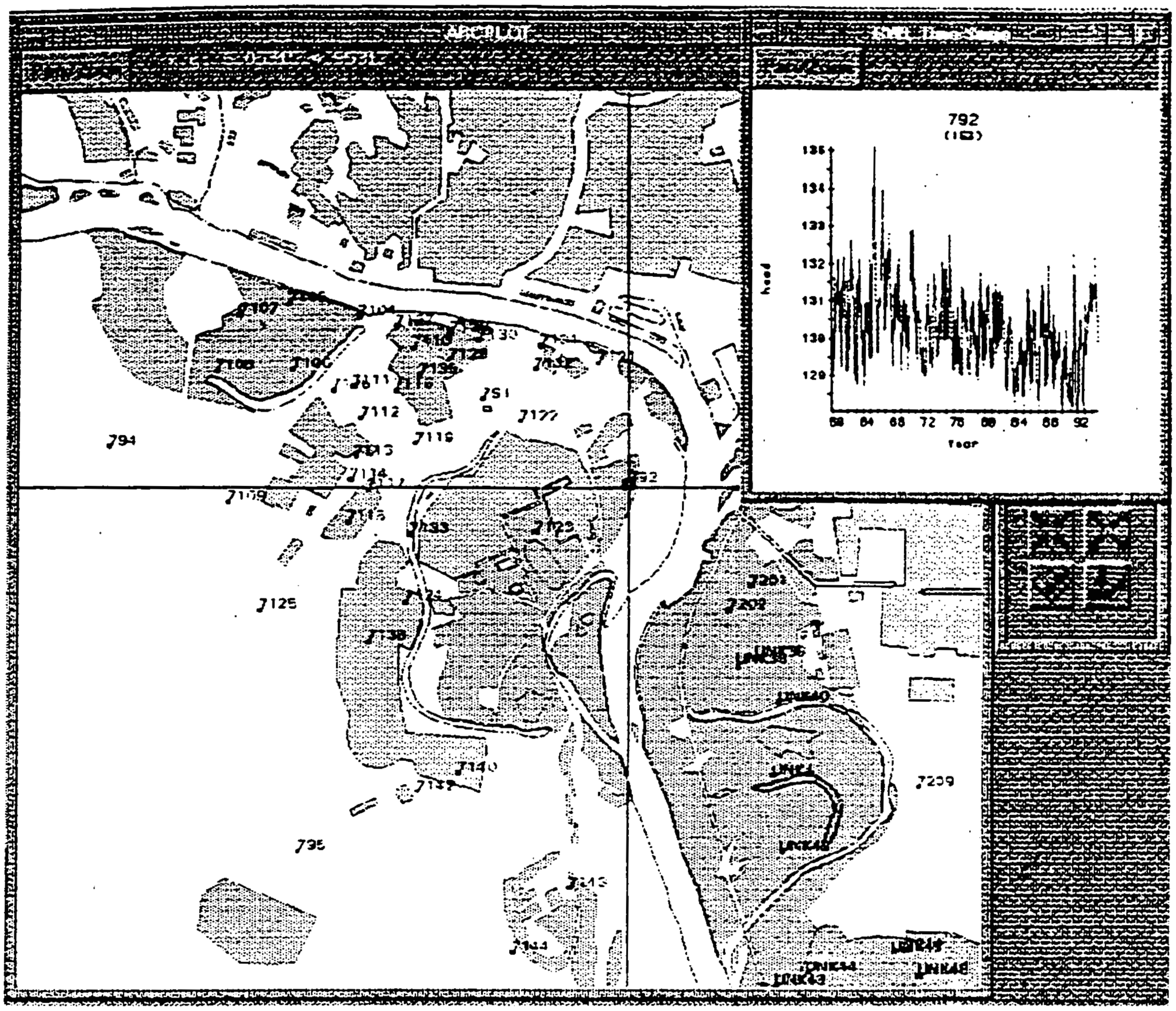

Fig. 2 Display of measured groundwater levels in DLIS. In addition the figure shows the Danube River and minor channels, location of groundwater IPTs and paved areas in the city of Bratislava (shaded).

create a desired lay out. In this phase DLIS has been used extensively.

MIKE 11-GIS

The MIKE 11-GIS is a fully menu-driven and generalized tool for flood mapping. The MIKE 11-GIS is linked to ARC/INFO GIS or ARC/VIEW GIS. MIKE 11-GIS was developed as part of the Flood Action Plan in Bangladesh. The project (FAP25) Flood Management Model was carried out by DHI. The overall objective of the study was to establish a tool which can improve flood management practice in Bangladesh.

Flood mapping with MIKE 11-GIS

The MIKE 11 modelling system provides a generalized tool for analysis, design, planning and forecasting of all aspects of river flood dynamics. By merging MIKE 11 with GIS technology a unique tool for displaying flood maps and related statistics is generated. Basically, flood maps are produced by comparing MIKE 11 results in terms of water elevations at different locations in rivers and on flood plains with a Digital Elevation Model (DEM). Two different types of flood map may be produced, namely flood depth and flood duration maps. In addition, comparison maps may be produced. Comparison maps show the difference between two flood depth maps or two flood duration maps. Hence, comparison maps are useful when studying results from two different scenarios, for instance in order to illustrate the difference in flood depth for a situation with and without flood protection measured such as embankments. A schematized flood depth map is shown in Fig. 3. MIKE 11-GIS includes facilities for producing different kinds of statistics, for instance maximum flood extent and time series of water level or discharge at different locations in the rivers. At present, a DEM is typically established by digitizing topographic maps, but in the near future satellite generated DEMs will be most probably be sufficiently precise.

Flood impact assessment with MIKE 11-GIS

By combining flood maps with other GIS coverage, flood impact assessment becomes possible. By combining different types of GIS coverage with flood maps a wide variety of flood impacts can be addressed by MIKE 11-GIS. A few examples are briefly described below.

Impacts on infrastructure can be assessed by combining flood maps with infrastructural GIS coverage. For instance, impacts on important transportation lines such as railways, roads and airports can be assessed.

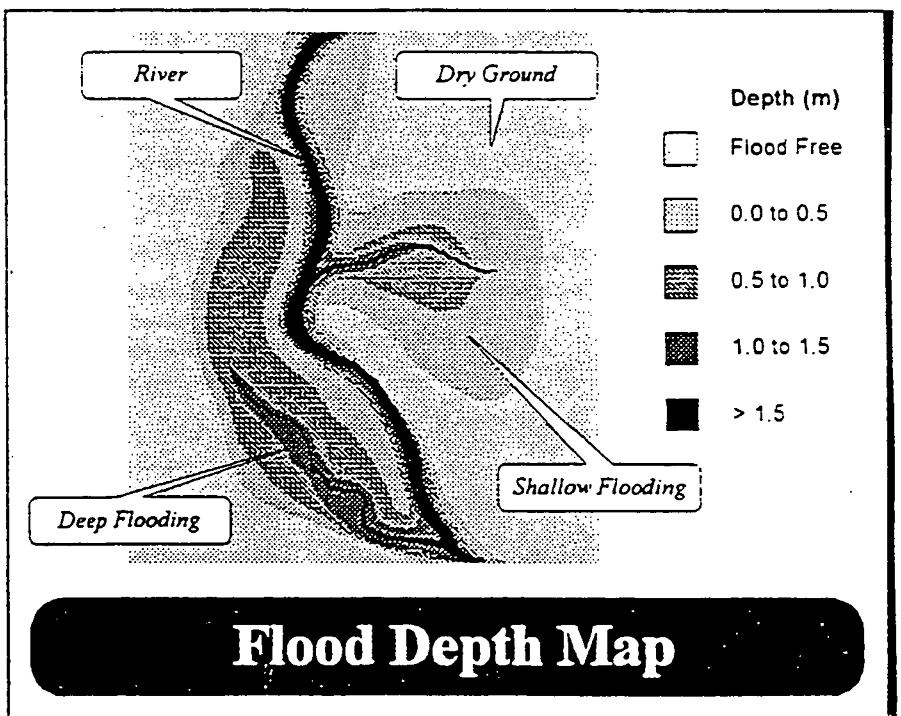

Fig. 3 MIKE11-GIS flood depth map.

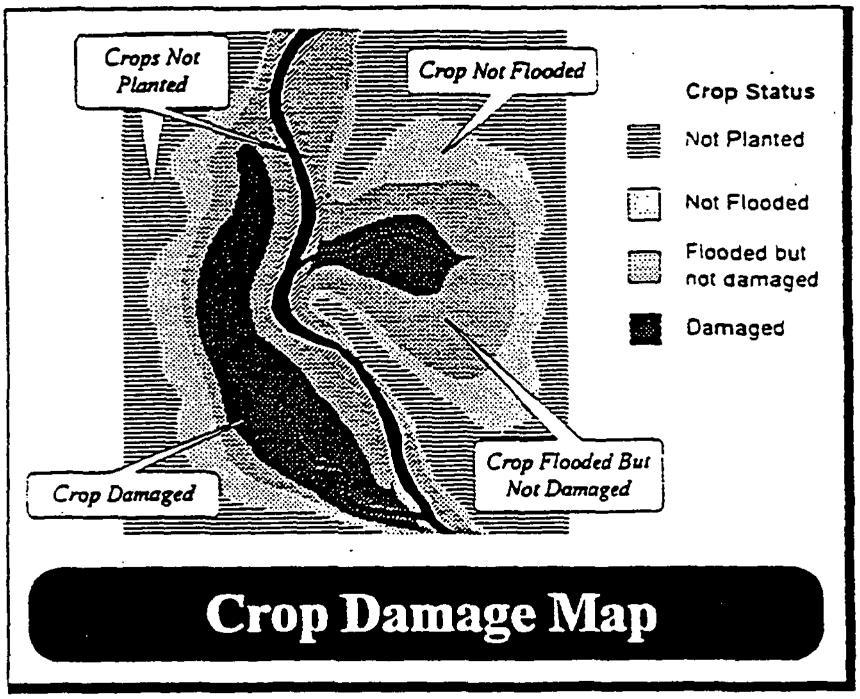

Fig. 4 MIKE11-GIS flood impact map.

Impact on communities can be assessed by combining maps of, for instance, population density with flood maps. In this regard, MIKE 11-GIS has been used as part of a flood forecasting tool applied by DHI to different regions of the world.

Environmental impacts can be assessed by combining flood maps with maps of, for instance, industries which store or produce highly toxic substances, or the location of landfills or special waste treatment plants.

Impacts on agriculture can be assessed by combining flood maps with coverage of different types of crops.

In order to make such flood impact assessments, some critical criteria expressed in terms of flood depth and/or flood duration must be established. For instance, short and shallow flooding of a certain crop may be beneficial for a crop while a too long or too deep flood may be lethal. For communities (people and animals), a long lasting shallow flood may not be critical while all deep floods may be lethal unless evacuation is carried out. A schematized flood impact map is shown in Fig. 4, illustrating the impacts of a flood event on a certain crop.

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CHAPTER 1
THE ROLE OF DISTRIBUTED HYDROLOGICAL MODELLING IN WATER RESOURCES MANAGEMENT

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1. Problems in Water Resources Management

"Scarcity and misuse of fresh water pose a serious and growing threat to sustainable development and protection of the environment. Human health and welfare, food security, industrial development and the ecosystems on which they depend, are all at risk, unless water and land resources are managed more effectively in the present decade and beyond than they have been in the past". (ICWE, 1992)

The present status and the future challenges facing hydrologists and water resources managers are summarized in this way in the introductory paragraph of the Dublin Statement on Water and Sustainable Development (ICWE, 1992). The Dublin Statement was adopted by government-designated experts from 114 countries and representatives of 80 international, intergovernmental and non-governmental organizations at the International Conference on Water and the Environment (a preparatory conference for the UNCED conference held in Rio de Janeiro in June 1992).

Since the ancient civilizations of Persia, Egypt and Babylon some 4000 years ago, water resources and water supply technology have played foundational roles in the development and organisation of many societies.

However, rapid population growth and the industrial development during the past few decades have caused an increasing pressure on land and water resources in almost all regions of the world. Due to increasing demands for water for domestic, agricultural, industrial, recreational and other uses and due to an increasing pollution of surface and groundwater, water resources have become scarce natural resources.

The availability of good-quality water is critical for human survival, economic development and the environment. Yet, water resources are not being managed in an efficient and sustainable manner. At the ICWE and UNCED conferences focus was put on past experiences of water resources management and new principles outlining improved future approaches were agreed upon. The World Bank in a follow-up policy paper (World Bank, 1993) emphasizes three problems which in particular need to be addressed:

* Fragmented public investment programming and sector management, that have failed to take account of the interdependencies among agencies, jurisdictions, and sectors
* Excessive reliance on overextended government agencies that have neglected the

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need for economic pricing, financial accountability, and user participation and have not provided services effectively to the poor

- Public investments and regulations that have neglected water quality, health and environmental concerns.

Central elements in the World Bank's new policy are adoption of a comprehensive policy framework and the treatment of water as an economic good, combined with decentralized management and delivery structures, greater reliance on pricing, and fuller participation of stakeholders.

Such new approach to water resources management requires, first of all, combined efforts of professionals from a large range of disciplines such as economists, administrators, engineers, hydrologists and ecologists, as well as a cross-sectoral integration in the planning and management process. As the traditions for cooperation and integration among these various disciplines and sectors have generally not been very strong, this challenge is very large and crucial.

Additionally, the increased water resources problems and the new management approach require improved water resources management tools based on sound scientific principles and efficient technologies. Key characteristics of such improved technologies are that they to a larger extent than the existing tools must facilitate a holistic view of water resources as well as cooperation among different disciplines and sectors involved in water resources management. This involves, amongst others, an integrated description of the entire land phase of the hydrological cycle, an integrated description of water quantity, quality and ecology, and integration of hydrological, ecological, economical and administrative information in information systems specifically designed for decisions makers at different levels.

The role of distributed hydrological models should be seen in this context. As will be described later in this chapter and in other chapters of this book distributed hydrological models are essential elements comprising some of these required capabilities. Hence, distributed hydrological models are important and necessary, but far from sufficient, tools in improving the future water resources management.

In Section 2 of this chapter a brief review is made of important present problems and trends related to water resources. Section 3 provides a review of the state-of-the-art in hydrological modelling aiming to assist in the analysis and management of these problems.

Section 4 contains a discussion on which factors limit the practical use of distributed hydrological modelling in water resources management.

2. Key Issues and Trends in Water Resources

2.1. EFFECTS OF EXPLOITATION OF WATER RESOURCES

In 1940 the total global water use was about 1,000 km^3 per year. It had doubled by 1960 and doubled again by 1990 (Clarke, 1991). In most countries of the world there is not enough readily available water of sufficient quality for another such doubling. Developments in some countries, such as China and India, suggest that this may be

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experienced within the first decade of the next century, unless some major improvements are made in water use efficiency.

One common result of a continuous exploitation of surface and groundwater sources is a periodic or permanent lowering of the groundwater table and/or water level in surface water bodies, thus limiting both the quantity and quality of water available for other users.

Heavy and sustained extraction of water and the resulting changes in its quantity, quality and accessibility can have irreversible effects on the flora and fauna in the affected area. For example, the Niger River dried out for the first time in history in 1986, due to the combined effects of drought and the expansion of areas under irrigation leading to increased water losses due to evaporation. Other major, regional examples are the Colorado River (Carrier, 1991) and the Aral Sea, which both have seriously deteriorated due to extensive water extraction in the upstream catchment area.

In addition to the lowering of the water table, other adverse effects of excessive groundwater abstraction include increased concentrations of pollutants in the aquifer due to reduced flow rates or changed geochemical conditions, increased risk of salt water intrusion and land subsidence. The latter phenomenon may be illustrated by Bangkok, where groundwater abstractions over decades for urban water supply has resulted in land subsidence, in some places by more than 10 cm per year, thus contributing to serious flooding (BMA, 1986).

2.2. IRRIGATION

The purpose of irrigation is to enable cultivation in regions, and during periods, otherwise unsuited for farming and to stabilize crop production in regions with large fluctuations in rainfall. Common sources of irrigation water are streams and rivers, downstream releases from reservoirs or pumping from groundwater aquifers. In addition, conjunctive use of surface and groundwater is particularly attractive in regions where dry season irrigation is impossible from surface water sources alone.

By the mid-1980s the irrigated agricultural land in the world amounted to approximately 220 million hectares. Only 15% of the world's crop land is irrigated but it contributes 30-40% of all agricultural production (FAO, 1990).

Irrigated agriculture accounts for about 70% of water withdrawals in the world, but the current overall performance of many irrigation systems is very poor. Inadequate operation and maintenance and inefficient management contribute to many environmental problems. In many irrigation schemes, 60% of the water diverted or pumped for irrigation is 'lost' on its way from the source to the plant. Hence, the potential for conserving water by increasing irrigation efficiency is tremendous.

The major environmental problems result from the excessive application of irrigation water to land with poor or non-existing drainage facilities. Under such conditions the groundwater table rises and the land finally becomes waterlogged with increasingly saline water and reduced crop yields. Once-thriving civilizations in such areas as Mesopotamia and ancient Sri Lanka were destroyed as a result of waterlogging and salinization, and the phenomenon is widespread today along the Indus, Nile, Tigris, Euphrates and in other semi-arid and arid regions of the world. According to FAO

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(1990), around 15% of the world's irrigated land is severely affected by salinity, and an additional 30%, approximately, are affected to some noticeable degree.

2.3. LAND DEGRADATION AND SOIL EROSION

Land degradation and soil erosion is another major problem worldwide. Soil erosion leads to the loss of valuable topsoil and causes silting, sedimentation, turbidity problems and pollution in downstream areas.

In Europe, for instance, erosion rates on agricultural land in the hilly areas of the Mediterranean and on sandy, loamy and chalky soils in northern Europe can reach 10-100 t/ha/year (Morgan et al, 1992). These rates should be compared with a value of 1 t/ha/year which is generally considered the maximum allowable for the control of pollution and the preservation of soil resources (Evans, 1981).

In some developing countries, land degradation is accelerated by increasing human and livestock populations, resulting in overgrazing, bushfires, exploitation of croplands and deforestation due to demand for firewood. In semi-arid and arid regions, such degradation is called desertification. According to FAO (1990), desertification affects nearly 75% of all productive rainfed lands and 60% of the rural population (280 million people) living is these areas.

Negligence and lack of knowledge of the importance of upland catchments in soil and water conservation do not only affect people living in these areas, but also result in considerable damage and losses for lowland populations due to the flooding and sedimentation of reservoirs. As a result, an estimated 160 million hectares of upland catchments in tropical developing countries have been seriously degraded, affecting approximately 20% of the world's population (Danida, 1988).

2.4. SURFACE AND GROUND WATER POLLUTION

Until a few decades ago water quality was relatively unimportant, except in arid lands where salinization occurred. Population growth, urbanisation and industrialization have now resulted in such increased water demands and such levels of contamination of water from the disposal of wastes that the use of water today is limited by its quality rather than the quantity available in many areas.

Major issues in surface water pollution are pathogenic agents, organic pollution, heavy metals, pesticides and industrial organics, acidification and eutrophication (WHO, 1991).

The extent and severity of the contamination of unsaturated zones and aquifers have been underestimated in the past due to the relative inaccessibility of the aquifers and the lack of reliable information on aquifer systems generally.

In many industrial countries groundwater pollution has become a key issue within the last two decades. For example in Denmark, where more than 99% of the water used is abstracted from groundwater, it was a general belief 15 years ago that the risk of groundwater pollution was small. Today, surveys have indicated the existence of more than 10,000 'hot spot' point sources for groundwater pollution in terms of landfills, chemical dumps, leaky oil tanks and many others and more than 400 million DKK (about 70 million US$) is spent annually for monitoring and remediation in order to save the country's groundwater resources for the future. Furthermore, nitrogen and pesticide pollutions due to agricultural activities have been recognized as very serious threats, and these will necessitate significant, and potentially very expensive, changes in agricultural practices, if these effects are to be reduced in the future.

2.5. FLOODS AND DROUGHTS

Economic losses from natural disasters increased three-fold between the 1960s and the 1980s (ICWE, 1992), while floods and droughts kill more people and cause more damage than do any other natural disasters (Rodda, 1995).

In spite of a continuous and ongoing construction of flood control structures such as reservoirs and dikes the flood damages in many countries have continued to increase. With an increasing pressure on land, also in flood plains, and an increasing awareness of the potential negative ecological impacts of large reservoirs and other regulation measures, these flood damages can be expected to increase significantly in the future. The recent (1993, 1994, 1995) very large floods in the continental rivers Mississippi and Rhine, both of which were considered to be 'well controlled', have generated considerable uncertainty about whether the hydrological regime has changed due to climate change or changes in land use, or both simultaneously.

2.6. AQUATIC ECOSYSTEMS

Water is a vital part of the environment and a home for many forms of life on which the well-being of humans also ultimately depends. Disruption of flows has reduced the productivity of many such ecosystems, devastated fisheries, agriculture and grazing, and marginalized the rural communities which rely on these. Various kinds of pollution exacerbate these problems, degrading water supplies, requiring more expensive water treatment, destroying aquatic fauna, and denying recreation opportunities (ICWE, 1992)

As an example, the destruction of wetlands has generally taken place at a dramatic rate during this century, and especially so because these were formerly viewed, quite wrongly, as wastelands. Thus, in the USA, 54% of the original wetlands had been lost by the mid-1970s (Tiner, 1984), and similar figures apply for several countries in Europe (Adams, 1986; Dugan, 1993).

2.7. POSSIBLE CLIMATE CHANGES

It is presently accepted that significant man-induced climate change may occur over the next decades, in particular related to the increase of CO_2 concentrations in the atmosphere.

Among the most important impacts of climate change will be its effects on the hydrological cycle and its associated water management systems. Hence, there is a danger that the resources of some areas will be reduced while others will suffer an increase in flood damage.

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3. Model Applications in Water Resources Management - State-of-the-Art

Many reviews of hydrological models and their applicability to various water resources problems exist in the literature, e.g. Stanbury (1986), Bowles and O'Connel (1988), De Coursey (1988), Mangold and Tsang (1991) and Feddes et al (1988). Existing model reviews generally focus on the scientific and to some extent the technological aspects. However, only a few reviews focus on the status of practical applications of models.

In a review prepared for the Commission for the European Communities (SAST, 1992) the state-of-the-art of the existing modelling techniques was characterised both with respect to the scientific and the technological status. Furthermore SAST (1992) provided an overview of the status of practical use of models. An updated version of this status is given in Table 1. For each of the potential fields of model application the present status has been qualitatively assessed, ranging from "practically no operational applications" to "standard professional tool in many regions of the world". Furthermore, the major constraints for practical applications of models within the various fields have been identified.

It is realized that the scientific and technological status of the various modelling systems presently applied at different institutions varies considerably. The status given in Table 1 refer to the state-of-the-art versions of modelling systems. Some of the statements in Table 1 are justified and elaborated in the following subsections 3.1 - 3.9.

3.1. WATER RESOURCES ASSESSMENT

Water resources assessment is the determination of the quantity, quality and availability of water resources, on the basis of which an evaluation of the possibilities for their sustainable development, management and control can be made.

Sound water resources assessment requires both access to good hydrological data and the application of suitable modelling techniques. In cases where the focus is concentrated completely on surface water or completely on groundwater the relevant modelling tools are often rainfall-runoff models of the lumped conceptual type or traditional two-dimensional groundwater models, respectively. In cases where surface water and groundwater interaction is important, more comprehensive modelling tools are required, such as distributed physically-based integrated catchment models.

In general, for whatever kind of application, adequate model codes exist and are being used in many cases. There is however a need and a potential for a significant increase in model application for water resources assessment. The main constraint in this respect is most often an administrative tradition. However, technological innovations such as improved user friendliness and computer-assisted parameter estimation methods are also important.

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TABLE 1. Status of application of hydrological modelling systems to various problem types

Field of Problem	STATUS OF APPLICATION				
	Adequacy of Scientific Basis	Scientifically Well Tested ?	Validation on Pilot Schemes ?	Practical Applications	Major Constraint for Practical Application
Water resources assessment					
• Groundwater	Good	Good	Adequate	Standard/Part	Administrative
• Surface water	Very good	Very good	Adequate	Standard/Part	Administrative
Irrigation	Good	Good	Partially	Very limited	Techno/Admin
Soil erosion	Fair	Fair	Very limited	Nil	Science
Surface water pollution	Good	Good	Adequate	Some cases	Administrative
Groundwater pollution					
• Point sources (landfills)	Good	Good	Partially	Standard/Part	Techno/Admin
• Non-point (agriculture)	Fair	Fair	Very limited	Very limited	
On-line forecasting					
• River flows/water levels	Very good	Very good	Adequate	Standard	Nil
• Surface water quality	Good	Good	Adequate	Standard/Part	Data/Admin
• Groundwater heads/w.table	Very good	Very good	Partially	Very limited	Data/Techno
• Groundwater quality	Fair	Fair	Nil	Nil	Science
Effects of land use change					
• Flows	Good	Fair	Fair	Very limited	Science
• Water quality	Fair	Fair	Fair	Nil	Science
Aquatic ecology	Fair	Fair	Very limited	Very limited	Science/Techno
Effects of climate change					
• Flows	Good	Good	Fair	Very limited	Science
• Water quality	Fair	Fair	Nil	Nil	Science

LEGEND:

Adequacy of scientific basis
- Poor: Large and crucial needs for improvements in scientific basis
- Fair: Considerable needs for improvements in scientific basis
- Good: Some needs for improvements in scientific basis
- Very good: No present significant need for improvements in scientific basis

Scientifically well tested ?
- Poor: Large needs for fundamental tests of scientific method
- Fair: Considerable needs for testing (some) of the scientific basis
- Good: Some needs for testing of the scientific basis
- Very good: No present significant need for testing of the scientific basis

Validation on pilot schemes ?
- Nil: No successful validation on well controlled pilot schemes so far - urgent need for validation on pilot schemes
- Very limited: A few (a couple of) validation cases - considerable needs for more validation projects on pilot schemes
- Partially: Some cases with successful validation on pilot schemes - some needs for further validations
- Adequate: Many good validation - no further present needs

Practical applications
- Nil: Practically no operational applications
- Very limited: A few well proven cases of operational practical applications
- Some cases: Some cases of well proven operational practical applications
- Standard/Part: Standard professional tool in some regions
- Standard: Standard professional tool in many regions of the world

Major constraint for further practical application
- Data: Data availability a major constraint
- Science: Inadequate scientific basis is a major constraint
- Technology: A technology push is required in order to make well proven methods more widely applicable
- Administrative: Administrative tradition or missing economical motivation is a major constraint

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3.2. IRRIGATION

The irrigation sector is, with few exceptions, dominated by a low technological level as far as hydrology is concerned. Thus, whereas the potential for improved irrigation management by use of modern technology is tremendous with regard to positive economic and environmental impacts, few attempts have been made so far to exploit this.

The requirements for the modernization of irrigation management include the following key elements:

* Improved data acquisition techniques including modern sensors, on-line data transmission and spatial information from remote sensing data.
* Detailed hydrological/hydraulic modelling enabling full descriptions of soil moisture and groundwater conditions on a spatially distributed basis in the command area as well as dynamic modelling of water flows and storages in the distribution and drainage channel system. One of the first attempts in this regard was that made by Lohani et al. (1993).
* Optimization techniques for managing the operation of reservoirs and other hydraulic control structures.

The scientific basis appears adequate, while the two major constraints are the administrative and engineering tradition in combination with a lack of well proven, fully integrated, and user friendly technological solutions.

3.3. SOIL EROSION

The soil and water conservation sector is also characterized by a low technological level as far as hydrological modelling is concerned. The most common method of estimating soil erosion from a catchment is still the 'Universal Soil Loss Equation (USLE)' (Wischmeier and Smith, 1965) which is a very simple, empirical equation originally developed for hand calculations (see also Lørup and Styczen, Chapter 6).

A number of physically-based soil erosion models are being developed but more research is required on process descriptions before large scale applications will be feasible.

3.4. SURFACE WATER POLLUTION

The state-of-the-art in surface water quality modelling is comparatively good both with regard to the scientific and technological status and models are being applied extensively.

3.5. GROUNDWATER POLLUTION

The main problem in studies of groundwater pollution relates to obtaining a sufficiently detailed three-dimensional description of the geology and in this way obtaining data on the spatial variability of the hydraulic parameters which ultimately determine the

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transport and spreading of contaminants (Hansen and Gravesen, Chapter 10). In general, the modelling technology with regard to groundwater flow and transport is well advanced in comparison to the data acquisition problem (Storm and Refsgaard, Chapter 4).

With regard to groundwater quality processes, research is still required on process identification and associated parameter assessment for organic and inorganic geochemical processes and their interactions (Engesgaard, Chapter 5). With regard to non-point agrochemical pollution, some basic process descriptions in the root zone still require research, especially with regard to the importance of agricultural management techniques, such as tillage (Thorsen et al., Chapter 7).

3.6. ON-LINE FORECASTING

Hydrological models in combination with modern on-line data acquisition systems are being used as standard tools for real-time flood forecasting purposes. At present, lumped conceptual rainfall-runoff models in combination with hydrodynamic river routing models represent the state-of-the-art for this type of application, and it is not likely that more sophisticated distributed physically-based models in general can provide significantly better levels of accuracy and overall reliability.

The scientific and, to a large extend, the technological basis also exist for applying on-line forecasting systems in the fields of surface water quality and groundwater flow and head estimation. In these areas the administrative tradition represents a constraint upon practical applications.

3.7. EFFECTS OF LAND USE CHANGE

Prediction of effects of land use change on water quantity and water quality represent an area of increasing importance. For instance, the possible effects of urbanisation or deforestation on floods and droughts and the possible effects of changed cropping pattern and other agricultural practises on soil erosion and ground water quality are key issues in water resources management in many areas (Lørup and Styczen, Chapter 6; Thorsen et al., Chapter 7).

Whereas the existing modelling tools are very useful in addressing these issues, the major constraint for widespread model applications in these subjects is a lack of basic knowledge on process descriptions and parameter values.

3.8. AQUATIC ECOLOGY

Modelling of wetlands and aquatic ecology has traditionally been dominated by very simple hydrological modelling tools, because the ecological processes themselves are extremely complex and data demanding and because the most advanced hydrological models, until recently, have not been able to provide sufficiently detailed descriptions for ecological modellers. Hence, comprehensive modelling of aquatic ecology has until now been carried out in relatively few cases.

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However, a prerequisite for establishing a predictive capability for management purposes within aquatic ecology is to make use of the most advanced of the distributed physically-based modelling systems. An example of a comprehensive floodplain model is given by Sørensen et al. (Chapter 12).

Thus, the main constraints presently are of scientific and technological nature. These constraints may be expected to be reduced as inverse methods, such as those effected by Kalman filtering, neural networks, and genetic algorithms are brought into regular use, while more advanced knowledge-based ecological models will no doubt also contribute further to advancing this area of application (e.g. Abbott et al., 1994).

3.9. EFFECTS OF CLIMATE CHANGE

Prediction of the hydrological effects of climate change is maybe one of the most difficult issues with which hydrology is confronted. With todays technology the impact of specified changes in precipitation, temperature and evaporation on river runoff, soil moisture regime and groundwater recharge can be calculated with reasonable accuracy, and any number of simple models can be constructed that will reproduce the one or the other aspect of climate change.

However, a climate change will, gradually, also result in successions of vegetation types, changes in agricultural practises etc, which may be expected to have significant effects on water resources. Thus, a considerable amount of further research is required on these issues.

A major weakness in the present generation of climate models is their very simple description of land surface processes, especially soil moisture and its spatial variations, which to a large extent control the land surface - atmosphere interactions. Distributed hydrological models appear to be suitable for this purpose, but have not been used so far maybe due to lack of interdisciplinary interactions.

3.10. HISTORICAL RECONSTRUCTIONS OF THE IMPACTS OF HUMAN ACTIVITY

A considerable interest accrues also to the reconstruction of the hydrology of river basins and other areas as these have changed due to changing land use practises and the construction of structures over considerable periods of time. Indeed, so extended are these periods that one can perhaps better speak of 'hydrological archaeology'. This is necessary to determine the causes and possible remedies for often catastrophic flood events for the purposes of risk assessment, insurance and investment planning, legislation and litigation.

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4. A discussion on Constraints for the Practical Use of Distributed Hydrological Modelling in Water Resources Management

4.1. THE NEED FOR DISTRIBUTED HYDROLOGICAL MODELLING

It is evident from Section 3 that there is a growing need for advanced distributed physically-based models. The traditional hydrological models of lumped conceptual type are well suited to deal with the main part of current water resources assessment and flood and drought forecasting, but more advanced tools are required for the remaining problems. It is noticed that for many of the problem areas the need for the distributed models reflects a demand for predictive capability on the effects of man-induced impacts. Thus, there is a growing need to use distributed models as a management tool.

In the current discussion on the role and capabilities of distributed models versus lumped models (Beven, Chapter 13A; Refsgaard et al., Chapter 13B) much focus is put on rainfall-runoff modelling, whereas it should be recognized that the main challenge and potential applications for distributed models lie much beyond this field of application - in more difficult areas.

4.2. CONSTRAINTS FOR APPLICATIONS OF DISTRIBUTED HYDROLOGICAL MODELS

Computer-based hydrological modelling has been carried out for more than three decades. However, as elaborated by Klemes (1988) in a discussion of the (lack of) scientific tradition in hydrology, the traditional deterministic hydrological models of lumped conceptual type, such as the Sacramento model, are technological tools which can not rightfully be claimed to be scientifically sound. Klemes (1988) generally questioned the reliability of predictions made by such models. Similarly, Abbott (1972) in a survey of hydrological models found little of predictive value.

The needs and the concept for a physically-based distributed catchment model were initially outlined by Freeze and Harlan (1969). With Freeze's pioneering work as a particular inspiration, three European organizations in 1976 started the development of the Système Hydrologique Européen (SHE) (Abbott et al., 1986a,b). Since then, a large number of other distributed models have been developed.

Nevertheless, in spite of two decades of modelling development, distributed hydrological models are today being used in practise only at a fraction of their potential. For example an initial marketing survey conducted for the Commission of the European Communities by the organisations behind the SHE in 1978 indicated a market in the order of 21 billion ECU for works that would benefit from the application of such modelling technology. With this background a distribution of the two present SHE versions, MIKE SHE and SHETRAN, to not more than some 60 organisations world-wide by the end of 1994, is far below the initial expectations of its developers. This naturally calls for an explanation. With our background in developments and applications of the SHE over the past two decades we shall present our perception of the main reasons, or the constraints, that have given rise to this slower-than-expected development.

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4.2.1. Data availability

A prerequisite for making full use of the distributed physically-based models is the existence and easy accessibility of a large amount of data, including detailed spatial information on natural parameters such as geology, soil and vegetation and man-made impacts such as water abstractions, agricultural practices and discharge of pollutants.

In many cases all such relevant data do not exist and even the existing data is most often not easily accessible due to the lack of suitably computerized data bases. A further complication in this regard is the administrative problem created by the fact that these models, in addition to the traditional hydrometeorological data, require and can make use of many other data sources, such as those arising from agricultural, soil science and geological investigations.

Thus, distributed models have had to work with such data as happens to be available, which are rarely if ever collected with a view to their compatibility with distributed hydrological modelling activities. Hence, most distributed model codes are able to run with different levels of data availability, and indeed in many cases (topographic data, vegetation maps, etc.) are the only means whereby this data can be introduced directly into decision-making processes.

For many years, high expectations have been directed to remote sensing techniques for providing spatial data of use in distributed hydrological models. However, so far operational use of remote sensing data are, with the exception of satellite-inferred snow cover data and land use/vegetation mapping, not common practise. With the launching of new satellites in recent years improved possibilities arise and, as discussed by De Troch et al. (Chapter 9), there are now good reasons to expect the long awaited break-through for large scale operational application of remote sensing data jointly with distributed hydrological models.

Another important development gradually improving the availability of data is the application of GIS technology, which is particularly suitable for couplings with distributed hydrological models (Deckers and Te Stroet, Chapter 11).

4.2.2. Lack in scientific-hydrological understanding

With the introduction of a new modelling paradigm and concurrent research in process descriptions, new shortcomings in the scientific-hydrological understanding have emerged, especially with regard to flow, transport and water quality processes at small scales and their up-scaling to describe larger areas. Some of the key scientific problems are highlighted in several chapters of this book.

These scientific shortcomings have, on the one hand, constrained the practical applications of distributed hydrological models and, on the other hand, the existence and application of such models have put a new focus on some of these problems, thus contributing to advances in the scientific-hydrological understanding.

4.2.3. Traditions in hydrology and water resources engineering

The distributed physically-based model codes such as the SHE constituted a 'quantum jump' in complexity as compared with any other code so far known in hydrology. Moreover, it used technologies, such as had been developed in computational hydraulics, with which few hydrologists were familiar. Although the numerical-algorithmic problems could be largely overcome by development of the codes into user-friendly fourth generation modelling systems with well proven algorithms, such as the MIKE SHE, the problem was then shifted back to one of comprehending the fully integrated complexity of the physical system that was being modelled together with the constraints that were inherent in the modelling procedure. Very few professional engineers and managers were, and still are, educated and trained with the necessary integrated view of hydrological processes in anything like their real physical complexity.

This difficulty is exacerbated by the very nature of hydrology itself, whereby most professionals posses only limited view on the totality of the physical processes. Soil physicists, plant physiologists, hydrogeologists and others usually have only a very partial view on the whole system, while there are few organisations that have available both the full range of such specialists and the more broader-ranging professionals that are needed in many situations to exploit the potential of distributed physically-based codes to such a degree that this exploitation is economically justified.

4.2.4. Technological constraint

In order to achieve a large dissemination of modelling technology to a considerable part of the professional community (and not only to experienced hydrological modellers) experience from hydraulic engineering shows that so-called fourth generation modelling systems (user-friendly software products) are required. Furthermore, it is believed that fifth generation systems are required in order for the modelling technologies to achieve their full potential in terms of practical application. The fifth generation systems are hydroinformatics-based including some of the elements outlined in Section 4.3. More exact definitions of fourth and fifth generation systems are given by Refsgaard (Chapter 2). At present only a few fourth generation distributed physically-based hydrological modelling systems exist and fifth generation systems are still at an experimental stage. Experience in hydraulics however, where more than 2000 organisations already make use of fourth generation systems, suggests that this situation will change in the foreseeable future.

4.3. THE FUTURE ROLE OF HYDROINFORMATICS

In order to ensure a better use of the existing (and coming), most scientifically advanced models for practical application by a wide group of professionals and managers it will be necessary to integrate the hydrological model codes with new hydroinformatics technologies including, amongst others, the following elements (see also Babovic and Minns (Chapter 14):

Standards for "open" modelling systems. The more widespread application of modelling technology necessitates the development of standards defining common user interfaces and model application interfaces, so that several modelling systems can be easily coupled for specific applications. For instance, a hydrological flow model could in this way be coupled with a soil erosion model and a river sediment transport model, developed at different institutions, without changing anything in the programme codes,

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i.e these should be mutually full compatible. The standards should also ensure full compatibility and easy use of other commonly used software, such as the various classes of data bases, and then especially Geographical Information Systems (GISs).

Logical modelling techniques. The application of logical modelling techniques, that is, the coupling of numerical and logical programming, as already well advanced in hydraulics, must be expected to have its own impact on the knowledge engineering aspects of hydrology.

Knowledge base systems. The application of proprietorial knowledge base systems for various fields within water resources management can be expected to spread correspondingly. Hence, knowledge elicitation technologies, making use of computer aided knowledge engineering/elicitation tools, will also need to be applied in this field also.

Systematic calibration. The application of methods for calibration of hydrological models on an objective basis making use of both the available data and the hydrologist's experience may be expected to advance further. This will unavoidably involve many elements of inverse modelling and introducing and combining many aspects of expert-systems.

Optimization methods integrated with advanced models. The application of optimization methods which are fully integrated both with advanced models and with the above indicated hydroinformatics tools can be expected to be developed in hydrology as they are now currently being applied in hydraulics.

Decision methods. The application of decision methods and the integration of such methods with hydrological modelling systems for use in water resources control and management will have also to be advanced. At this stage the hydrological model becomes integrated in the new kinds of architectures and paradigms (of object orientation and agent orientation) that are now becoming established in other fields.

The ultimate output of such research activities is a fifth generation modelling environment, which may be described as a *virtual hydrological environment*. It will make possible a combined access to several powerful features such as:

- standard interfaces to the most common data bases and GIS's, giving the user access to necessary specific data,
- a choice of alternative and compatible hydrological modelling systems,
- graphical interface,
- utilities for model calibration, optimization, decision making and other methods, and
- enable professionals from other fields (agronomists, ecologists, meteorologists, etc) to access integrated hydrological data and knowledge resources in an efficient and responsible way.

The declarations adopted by the ICWE-Dublin and the UNCED-Rio conferences call for the abandoning of traditional sectoral approaches and the adoption of more integrated water resources management strategies. As stated by Matthews (1994) in a description of the World Bank's new policy, the need for "use of hydroinformatics for the implementation of the Dublin and Rio declarations becomes evident".

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CHAPTER 2
TERMINOLOGY, MODELLING PROTOCOL AND CLASSIFICATION OF HYDROLOGICAL MODEL CODES

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1. Introduction

All hydrological models are simplified representations of the real world. Models can be either physical (e.g. laboratory scale models), electrical analogue or mathematical. The physical and analogue models have been very important in the past. However, the mathematical group of models is by far the most easily and universally applicable, the most widespread and the one with the most rapid development with regard to scientific basis and application. The present book is devoted entirely to mathematical models.

The present book deals with simulation models in contrary to optimization models. In Section 2 a consistent, general terminology terminology and methodology applicable within the whole range of hydrological modelling is presented. Important elements related to the methodology are described within the modelling context. Alongside this, a terminology that is more suited to hydroinformatics applications, where such process models form only part of more general information and knowledge-based systems, is also introduced.

To a user, a hydrological model is composed of two main parts, namely a hydrological core and a technological shell. The hydrological core is based on a certain hydrological scientific basis providing the definitions of variables, the process descriptions and other aspects. The technological shell is the programming, user interface, pre- and postprocessing facilities etc. These two different and equally important aspects are addressed in Section 3 and Section 5, respectively. Section 4 is devoted to the problems caused by spatial variability of hydrological parameters and the fundamental different ways that different model types takes this aspect into account.

2. Terminology and Methodology

2.1. DEFINITIONS OF BASIC TERMS

No unique and generally accepted terminology is presently used in the hydrological community. Therefore, definitions of some of the most common terms used in hydrological modelling and in hydroinformatics and employed in the following text are given here.

The *natural system* that is considered here is the hydrological cycle or parts of it as

M. B. Abbott and J. C. Refsgaard (eds.), Distributed Hydrological Modelling, 17–39.

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we currently conceive it. This conception is naturally a function of our own social environment, historical developments, linguistic traditions and other variables that influence all disciplines, and not just hydrology. We shall suppose however that this process of conceptualisation is uniform over current hydrological practice. Thus, to use language of computer science that is also adopted by hydroinformatics, we shall assume that there is 'one universe of discourse', and that this is uniform.

A *hydrological model* is a simplified representation of the natural system. From a hydroinformatics point of view, a model is a collection of signs that serves as a sign, so that the hydrological model is the set of signs (symbols and other tokens) that serve as a representation of the natural system or some aspects of it.

A *mathematical model* is a set of mathematical expressions and logical statements combined in order to simulate the natural system. This is then, hydroinformatically, a model in which the signs are symbols organised within definite formal systems called mathematical languages.

Simulation is the time-varying description of the natural system computed by the hydrological model. A simulation may be seen as the model's imitation of the behaviour of the natural system. In hydroinformatics we speak of 'virtual worlds' that provide signs that mirror (albeit possibly in a simplified or distorted way) the signs that we ourselves experience as 'the world of nature'. Our 'time consciensness' can then also be mirrored, whether by a succession of images or through other, static-graphical measures.

A *routine*, a *component*, or a *submodel* is part of a more comprehensive model, e.g. the snowmelt simulation part of a model for the complete land phase of the hydrological cycle. Hydroinformatics speaks, with computer science generally, of *an object*, as a representation of anything to which our thoughts can be directed. When an object encapsulates knowledge that acts upon information so that it both acts upon and is itself acted upon by other objects, it is called an *agent*.

A *parameter* is a constant in the mathematical expressions or logical statements of the mathematical model. It remains constant in virtual time. A *variable* is a quantity which varies in space and time. It can be a series of inputs to and outputs from the model, but also a description of conditions in some component of the model. In hydroinformatics it is an indicative sign (in the sense of Husserl), the indication of which is towards a state of affairs that varies in time.

A *deterministic model* is a model where two equal sets of input (i.e. collections of signs) always yield the same output sign if run through the model under identical conditions. A deterministic model has no inner operations with a stochastic behaviour.

A *stochastic model* has at least one component of random character which is not explicit in the model input, but only implicit or 'hidden'. Therefore, identical inputs will generally result in different outputs if run through the model under, externally seen, identical conditions.This notion can be extended to models in which the input has a direct stochastic character.

We conceive a *catchment* as a region of physical space over which flows occur that are collectively of concern to us. A catchment is thus an object towards which the universe of discourse of hydrology becomes directed.

A *lumped model* is a model where the catchment is regarded as one unit. The

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variables and parameters are thus representing average values for the entire catchment. Thus the virtual world is reduced to just one object.

A *distributed model* take account of spatial variations in all variables and parameters.

By a *physical process* we understand a representation in our own minds of an event occurring in our outer world of sense experience.

A *black box* or an *empirical model* is a model developed without any consideration of the physical processes that we otherwise associate with the catchment. The model is merely based on analyses of concurrent input and output time series.

A *conceptual model* is one that is constructed on the basis of the physical processes that we 'read' into our observations of the catchment. In a conceptual model, physically sound structures and equations are used together with semi-empirical ones. However, the physical significance is not usually so clear that the parameters can be assessed from direct measurements. Instead, it is necessary to estimate the parameters from calibrations, applying concurrent input and output time series. A conceptual model, which is usually a lumped-type model, is often called a *grey box model*.

A *physically-based model* describes the natural system using the basic mathematical representations of the flows of mass, momentum and various forms of energy. For catchment models, a physically-based model in practice also has to be fully distributed. This type of model, also called a *white box model*, thus consists at its most basic 'human-friendly' level of a set of linked partial differential, integral-differential and integral equations together with parameters which, in principle, have direct physical significances and can be evaluated by independent measurements.

In connection with the technological classification, a distinction is made between a model and a modelling system. A *model* is defined as a particular hydrological model established for a particular catchment. A *modelling system*, on the other hand, is defined as a generalized software package, which, without program changes, can be used to establish a model with the same basic types of equations (but allowing different parameter values) for different catchments. The term *model code* is often used synonymously with the term modelling system. Thus, most of the models referred to in the present book are in fact generated using modelling systems. However, as this technological distinction between model and modelling system is not recognized rigorously throughout the hydrological scientific community, it is not maintained strictly in this book either.

2.2. GENERAL TERMINOLOGY FOR MODEL CREDIBILITY

Hydrological models are being developed and applied in increasing number and variety. At the same time contradictions are emerging regarding the various claims of model applicability on the one hand and the lack of validation of these claims on the other hand. Hence, the credibility of the advanced models can often be questioned, and often with good reason.

Many different definitions of model validation are presently used. A consistent terminology with a set of definitions for terms such as conceptual model, computerized model, verification, validation, domain of applicability and range of accuracy is given by Schlesinger et al. (1979). Slightly different definitions are used by other authors e.g.

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Konikow (1978), Tsang (1991), Flavelle (1992) and Anderson and Woessner (1992).

Oreskes et al. (1994), using a philosophical framework, states that verification and validation of numerical models of natural systems is theoretically impossible, because natural systems are never closed and because model results are always non-unique. Instead, in this view models can only be confirmed.

The following terminology is based on the general terminology proposed by Schlesinger et at. (1979), but is relativised by references to current usage in hydroinformatics (e.g. Abbott, 1991, 1993, 1994). The elements in the terminology and their interrelationship are illustrated in Fig. 1.

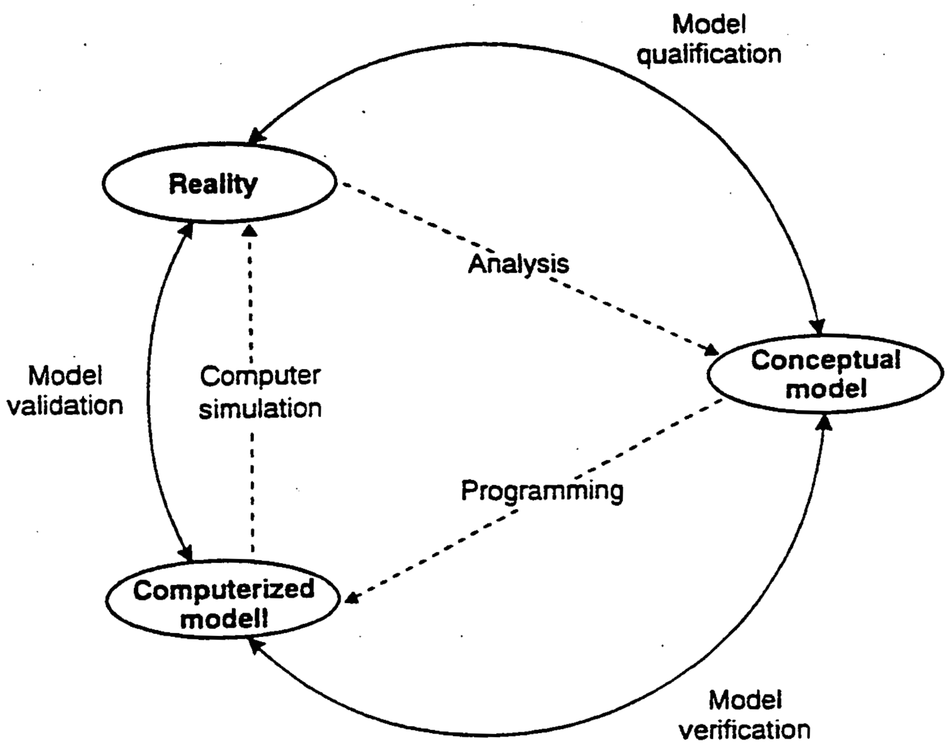

Figure 1. Elements of modelling terminology and their interrelationships. After Schlesinger et al. (1979).

Schlesinger et al. (1979) defines the different terms in a manner that appears the most suited to current hydrological thinking, while hydroinformatics is obliged to define them more generally, as follows:

Reality: The natural system, understood here as the hydrological cycle or parts of it. For hydroinformatics (as in philosophy and theology) reality and truth must be defined together. Thus (Abbott, 1994): "*Reality* is the name that we give to the interface between our outer and our inner worlds and a *truth* is an intimation of the oneness of these two worlds". Here our outer world is the world given to us by our senses and our inner world is our world of knowledge, including perceptual knowledge.

Conceptual model: Verbal descriptions, equations, governing relationships, or 'natural laws' that purport to describe reality. This is then the sign representation of that

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place where our outer and inner worlds intersect.

Domain of intended application (of a conceptual model): Prescribed conditions for which the conceptual model is intended to match reality. This is formulated differently in hydroinformatics, where reality in this sense is considered dynamic, and hence this domain is viewed as that to which the intentions of the user of the model is directed.

Level of agreement (of the conceptual model): Expected agreement between the conceptual model and reality, consistent with the domain of intended application and the purpose for which the model was built. This is often expressed in terms of performance criteria. This then becomes the observed agreement that can be expressed between the virtual world that the mathematical model is capable of reproducing when translated into code and the world of nature as we perceive it.

Model qualification: An estimation of the adequacy of the conceptual model to provide an acceptable level of agreement for the domain of intended application.

Computerised model: Anyone of an (in principle) infinite number of operational computer programs which implements a given conceptual model.

Model verification: Substantiation that a computerized model is in some sense a true representation of a conceptual model within certain specified limits or ranges of application and corresponding accuracy.

Domain of applicability (of a computerised model): Prescribed conditions for which the computerised model has been tested, i.e. compared with reality to the extent that is practically possible and judged suitable for use (through the process of model validation, described below).

Range of accuracy (of computerised model): Demonstrated agreement between the computerised model and reality within a stipulated domain of applicability. Since the introduction of Computational Hydraulics (e.g. Abbott, 1979) this has also been called the 'performance envelope' of the computerised model.

Model validation: Substantiation that a computerised model within its domain of applicability possesses a satisfactory range of accuracy consistent with the intended application of the model.

Although hydroinformatics can maintain some more or less tenuous relations to current practice in hydrology up to this point, as illustrated by the above definitions, at this point on it proceeds along a different track. As it makes recourse to earlier and longer established definitions, it treats *validation* in the manner set down by the Scholastics of the XIIth and XIIIth centuries and as carried over in the modern existentialistic movement by Kierkegaard. Modifying this point of view to suit the present context, there is a relation between ourselves and our outer world and another relation between our model and its outer world. The first is the relation established by our own sense experiences of the world of nature and the second is the relation expressed through the virtual world that the model presents to us. The process of validation then corresponds to a working out of the relation between these two relations so as to bring them as closely as possible into harmony (see Abbott, 1991), as schematised in Fig. 2

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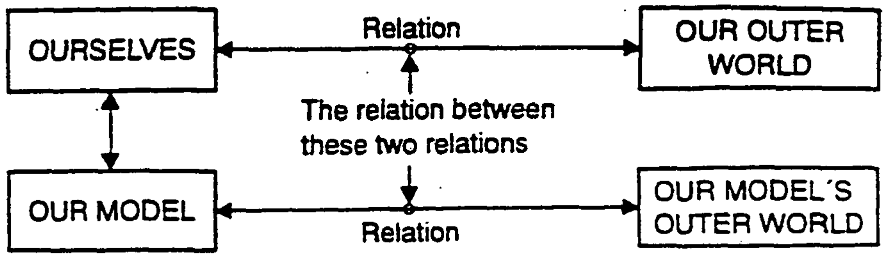

Figure 2. Schematic illustration of the term validation in traditional hydroinformatics sense as working out the relation between two relations.

It is generally understood, whether from theology, anthropology, mathematical logic or whatever other discipline that investigates this process, that this working out process is unending (e.g. Abbott, 1994). Thus models generally can only be more or less validated, but never 'absolutely validated'. This is to say that 'reality' and 'truth' can never coincide in any human construction.

Certification documentation: Documentation intended to communicate knowledge and information concerning a model's credibility and applicability, containing, as a minimum, the following basic elements:

(1) Statements of the purposes for which the model has been built.
(2) Verbal and analytical descriptions of the conceptual model and the corresponding computerised model.
(3) Specification of the domain of applicability and range of accuracy related to the purpose for which the model is intended.
(4) Description of tests used for model verification and model validation and a discussion of their adequacy.

Model certification: Acceptance by the model user of the certification documentation as adequate evidence that the computerised model can be effectively used for a specific application.

Computer simulation: Exercise of a tested and certified computerized model to 'gain insight into' reality or rather, in the general terms of computer semiotics (e.g. Anderson, 1990), to modify the user's perception of reality.

In the above definitions the term conceptual model should not be confused with the word conceptual used in the traditional classification of hydrological models ("lumped, conceptual" rainfall-runoff models), cf Subsection 2.1 and Section 3.

In practice the computerised model is usually not programmed separately for every case, but most often prepared on the basis of a generalised software package. It is therefore important to distinguish between the terms modelling system and model, as

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defined in Subsection 2.1 and Section 5.

In this context it should be observed that a modelling system or a code can itself be verified. A code verification involves comparison of the numerical solution generated by the code with one or more analytical solutions or with other numerical solutions. Verification ensures that the computer program solves the equations that constitute the mathematical model with an accuracy that is deemed adequate for the proposed application.

Similarly, a model is said to be validated if its accuracy and predictive capability throughout the process of validation has proven to lie within acceptable limits or errors. It is important to notice that the term 'model validation' refers to a site specific validation of a model. This must not be confused with a more general validation of a generalised modelling system, which in principle is not possible.

2.3. MODELLING PROTOCOL

There are numerous pitfalls into which the modeller can fall when using hydrological models. In this context it is essential that the user both has a thorough knowledge of the hydrological processes being modelled and has a solid experience in modelling. A third crucial factor is a good and rigorous procedure for applying models. Such a procedure, which comprises a sequence of steps in a hydrological model application, is often referred to as a modelling protocol.

In principle, such a protocol should be flexible, adaptive, open to new insights and, even when it becomes formalised as a specific list of actions, quite 'opportunistic' in its application. The protocol described below is a translation of the general terminology and methodology defined in Subsection 2.2 into the field of hydrological modelling. It is furthermore inspired by the modelling protocol suggested by Anderson and Woessner (1992), but modified concerning certain steps.

The protocol is illustrated in Fig. 3 and described step by step in the following. For each step the *reference to Fig. 3 are shown with italic text, [while the references to the general, current hydrological terminology in Fig. 1 and Subsection 2.2 are shown in brackets with italic text].*

(1) The first step in a modelling protocol is to *define the purpose* of the model application. Examples of purposes are rainfall-runoff simulation in a gauged catchment, prediction of changes in runoff pattern due to changes in land use, and the prediction of migration of contaminants. Other examples include interpretation of flow and transport pattern as a framework for assembling and organizing field data and formulating ideas about system dynamics. An important element in this step is to give a first assessment of the desired accuracies of the model outputs. Once the purpose has been sufficiently clearly defined, it may be obvious which type of modelling system is required in order to solve the specific problem. Of course it may happen that the results of the study indicate other possibilities and the possible need to use other tools, so that it may be necessary to return to this step during any one project. *[According to the terminology outlined in Subsection 2.2 this step corresponds to defining the domain of intended application.]*

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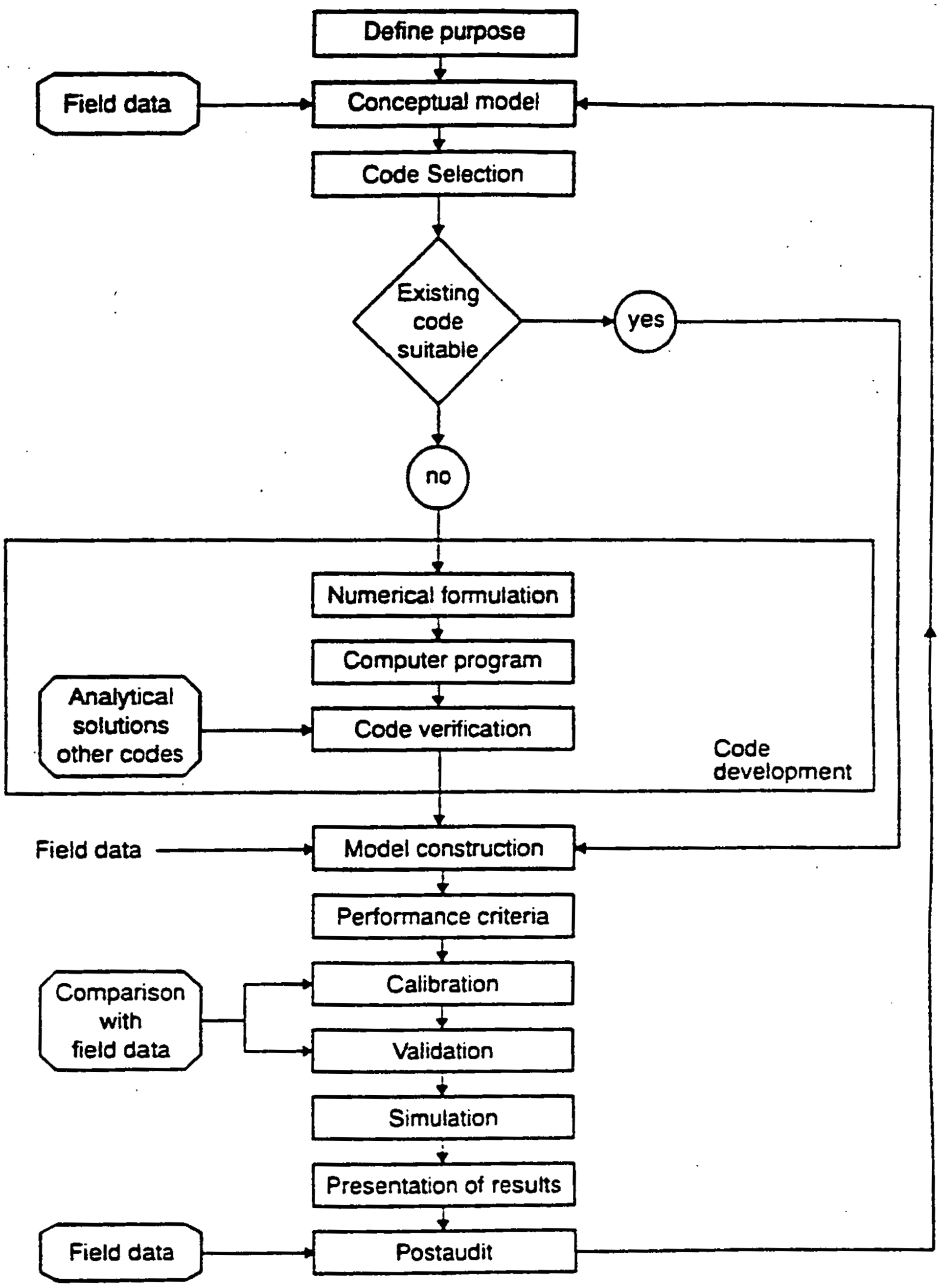

Figure 3. The different steps in hydrological model application - a modelling protocol. Modified after Anderson and Woessner (1992).

(2) Based on the purpose of the specific problem and an analysis of the available field data, the user must establish a *conceptual model*. In case of groundwater modelling this may, amongst others, involve investigating the geological conditions and decide the range of complexation to be included in the geological model. In the case of combined groundwater and surface water hydrology studies, it may, for instance, involve assessments of whether processes such as macropore

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flow and hysteresis in the unsaturated zone are important and need to be explicitly modelled. In other words, a conceptual model comprises the user's perception of the key hydrological processes in the catchment and the corresponding simplifications and numerical accuracy limits which are assumed acceptable in the mathematical model in order to achieve the purpose of the modelling. *[According to the terminology outlined in Fig. 1, this step corresponds to preparing a conceptual model on the basis of an analysis of the current perception of reality.]*

(3) After having defined the conceptual model, a suitable computer program has to be selected. In principle, the computer program can be prepared specifically for the particular purpose. In practice, a *code* is often *selected* among existing generalized modelling systems. In this case it is important to ensure that the selected code has been successfully verified for the particular type of application in question. *[The terminology in Subsection 2.2 does not explicitly consider selection of an existing code, but rather programming which is development of a new code.]*

(4) In case no existing code is considered suitable for the given conceptual model a a *code development* has to take place. The computer code is a computer program that contains an algorithm capable of solving the mathematical model numerically. The term computer code is here used synonymously with the term generalised modelling system. In order to substantiate that the code solves the equations in the conceptual model within acceptable limits of accuracy a *code verification* is required. In practise, code verification involves comparison of the numerical solution generated by the model with one or more analytical solutions or with other numerical solutions. *[According to the terminology outlined in Fig. 1, this step comprises programming and model verification.]*

(5) After having selected the code and collected the necessary field data, a *model construction* has to be made. This involves designing the model with regard to the spatial discretisation of the catchment, setting boundary and initial conditions and making a preliminary selection of parameter values from the field data. In the case of distributed modelling, the model construction involves parametrisation. This process is described in more details in Chapter 3 of this book. *[This step, together with step 7 below, corresponds to the process of establishing a computerised model referred to in Fig. 1.]*

(6) The next step is to define *performance criteria* that should be achieved during the subsequent calibration and validation steps. When establishing performance criteria, due consideration should be given to the accuracy desired for the specific problem (as assessed under step 1) and to the realistic limit of accuracy determined by the field situation and the available data (as assessed in connection with step 5). If the performance criteria are specified unrealistically high, it will either be necessary to modify the criteria or to collect more and possibly quite other field data. *[According to the hydrological terminology outlined in Section 2.2, this step corresponds to specifying the level of agreement.]*

(7) *Model calibration* in general involves manipulation of a specific model to reproduce the response of the catchment under study within the range of accuracy specified in the performance criteria. In practice this is most often done by trial-

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and-error adjustment of parameters, but automatic parameter estimation methods may also sometimes be used. It is important to assess the uncertainty in the estimation of model parameters, for example from sensitivity analyses. These calibration techniques are discussed further in Chapter 3 of this book. *[This step, together with step 5 above, corresponds to the process of establishing a computerised model referred to in Fig. 1.]*

(8) *Model validation* is the process of demonstrating that a given site specific model is capable of making sufficiently accurate predictions. This implies the application of the calibrated model without the changing of the parameter values that were set during the calibration when simulating the response for another period than the calibration period. The model is said to be validated if its accuracy and predictive capability in the validation period have been proven to lie within acceptable limits or to provide acceptable errors (as specified in the performance criteria). Validation schemes for different purposes are outlined in Chapter 3 of this book. *[It is this step that is also referred to as model validation in Fig. 1.]*

(9) *Model simulation* for prediction purposes is often the explicit aim of the model application. In view of the uncertainties in parameter values and, possibly, in future catchment conditions, it is advisable to carry out a predictive sensitivity analysis to test the effects of these uncertainties on the predicted results. *[This is referred to as computer simulation in Fig. 1.]*

(10) *Results* are usually *presented* in reports. However, with the newest information technology that is now emerging it is also possible to display modelling results as (colour) animations. Furthermore, in certain cases, the final model is transferred to the end user for subsequent day-to-day operational use. *[This step is not explicitly included in the general terminology in Section 2.2.]*

(11) An extra possibility of validation of a site specific model is a socalled *postaudit*. A postaudit is carried out several years after the modelling study is completed and the model's predictions can be evaluated. Examples of postaudits from groundwater modelling are given in Konikow (1986), Alley and Emery (1986) and Konikow and Person (1985). *[This step is not included in the general terminology in Section 2.2.]*

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3. Classification according to Hydrological Process Description

3.1. CLASSIFICATION

Many attempts have been made to classify hydrological models, see e.g. Fleming (1975) and Woolhiser (1973). The present classification, illustrated in Fig. 4. is applicable to hydrological simulation models. The classification is in principle applicable both to catchment models and to single component models such as groundwater models. It is emphasized that the classification is schematic, and that many model codes do not fit exactly into one of the given classes. The construction of such a classification corresponds to the introduction of a taxonomy into the hydrological universe of discourse.

The two classical types of hydrological models are the deterministic and the stochastic. During the 1960s and 1970s these two fundamentally different approaches were developed within two more or less separate "hydrological schools". However, over the last decade, increasingly more interplay has occurred and a joint stochastic-deterministic methodology today provides a very useful framework for addressing some of the fundamental problems in hydrology such as taking spatial variability (scale problems) into account and assessing uncertainties in modelling.

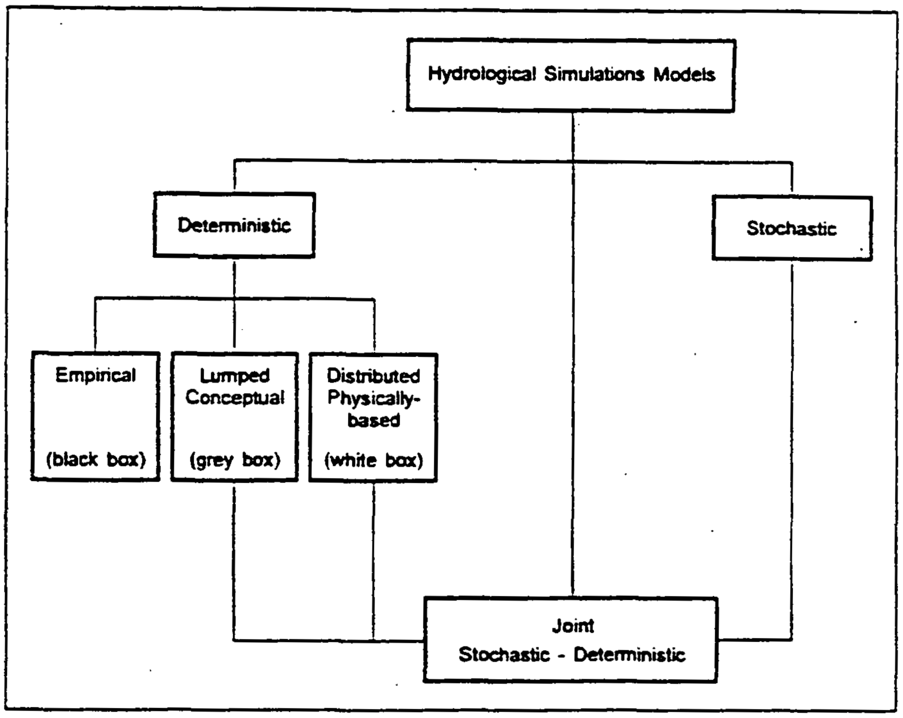

Figure 4. Classification of hydrological models according to process description.

3.2. DETERMINISTIC MODELS

Deterministic models can be classified according to whether the model gives a lumped or a distributed description of the considered area, and whether the description of the hydrological processes is empirical, conceptual, or more physically-based. As most conceptual models are also lumped, and as most physically-based models are also distributed, the three main groups of deterministic models shown in Fig. 4 emerge:

* Empirical models (black box).
* Lumped conceptual models (grey box).
* Distributed physically-based models (white box).

3.2.1. Empirical (black box) models

Black box models are empirical, involving mathematical equations that have been assessed, not from the physical processes in the catchment, but from analyses of concurrent input and output time series. Black box models may be divided into three main groups according to their origin, namely empirical hydrological methods, statistically based methods and hydroinformatics based methods.

Empirically hydrological methods. Probably, the best known among the black box models in hydrology is the unit hydrograph model and the models applying the unit hydrograph principles, Sherman (1932), Nash (1959). Today, empirical hydrological methods are often used in some components of more comprehensive models, e.g. the unit hydrograph is often used for streamflow routing and a linear reservoir is often used to represent the groundwater system in conceptual rainfall-runoff models.

Statistically based methods. Traditional statistical methods in hydrology have been developed extensively with support from basic statistical theory. These methods are often mathematically more advanced than the above empirical hydrological methods.

An important type of statistically based methods is comprised of the regression and correlation models. Linear regression and correlation techniques are standard statistical methods used to determine functional relationships between different data sets. The relationships are characterized by correlation coefficients and standard deviations and the parameter estimation is carried out using rigorous statistical methods involving tests for significance and 'validity' of the chosen model. Regression and correlation models are often used as so-called 'transfer function models' converting input time series to output time series. Examples of such models include:

* Autoregressive Integrated Moving Average (ARIMA) models (Box and Jenkins, 1970), which amongst others have been used extensively in surface water hydrology for establishing relationships between rainfall and discharge.
* The Constrained Linear Systems (CLS) model (Todini and Wallis (1977), which is basically a composition of different linear regression relationships valid for intervals between certain thresholds, so that it altogether functions as a non-linear model.
* Gauge to gauge correlation methods (e.g. WMO, 1994), which previously have constituted the most used method for flood forecasting in large rivers, e.g. in India and Bangladesh.
* Antecedant Precipitation Index (API) model (e.g. WMO, 1994) correlating rainfall volume and duration, the past days' rainfall, and the season of the year to runoff.

Hydroinformatics based methods. A new group of 'transfer function models' based on

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methods introduced more generally in hydroinformatics is now emerging. One class of techniques is based on neural networks, while another class is based on evolutionary algorithms. Both methods have been successfully tested on a trial basis for rainfall-runoff modelling, but they have not yet been sufficiently tested to be applied in practice. However, the potential for such methods appears to be significant, especially as a replacement of the statistically based methods, and for this reason they are introduced in Chapter 14 of this book.

3.2.2 Lumped conceptual models

Models of the lumped conceptual type are mainly found within the field of rainfall-runoff modelling. Lumped conceptual models operate with different but mutually-interrelated storages representing physical elements in a catchment. The mode of operation may be characterized as a bookkeeping system that is continuously accounting for the moisture contents in the storages.

Due to the lumped description, where all parameters and variables represent average values over the entire catchment, the description of the hydrological processes cannot be based directly on the equations that are supposed to be valid for the individual soil columns. Hence, the equations are semi-empirical, but still with a physical basis. Therefore, the model parameters cannot usually be assessed from field data alone, but have to be obtained through the help of calibration.

The Stanford modelling system (Crawford and Linsley, 1966) is the classical representative of this model type. The structure of the Stanford system is shown schematically in Fig. 5.

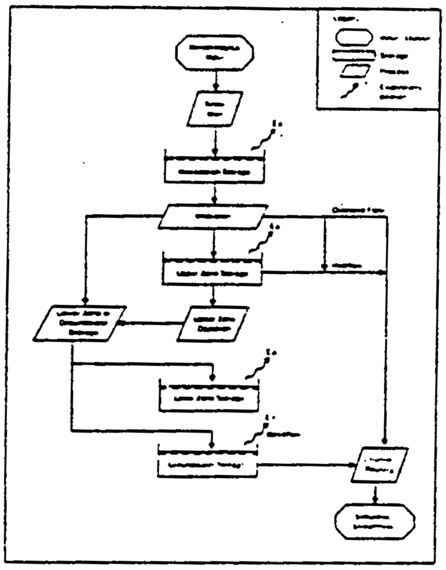

Figure 5. Structure of the Stanford model

Numerous other rainfall-runoff modelling systems of the lumped conceptual type exist. A brief description of 19 different systems is given by Fleming (1975). More comprehensive and recent descriptions of a large number of modelling systems are

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provided in Singh (1995).

The lumped, conceptual models are especially well suited for the simulation of the rainfall-runoff process when hydrological time series exist that are sufficiently long for a model calibration. Thus, typical fields of application include the extension of short streamflow records based on long rainfall records and real-time rainfall-runoff simulations for flow forecasting.

Although no significant improvements have been made during the last decade with regard to the fundamental structure and functioning of these models, many of the codes have undergone a comprehensive technological development (see Section 4) enabling them to be disseminated widely for practical application by a large group of users.

3.2.3 Distributed physically-based models

The principal mode of operation of a distributed physically-based model is illustrated in Fig. 6. Contrary to the lumped conceptual models, a distributed physically-based model does not consider the water flows in an area to take place between a few storage units. Instead, the flows of water and energy are directly calculated from the governing continuum (partial differential) equations, such as for instance the Saint Venant equations for overland and channel flow, Richards' equation for unsaturated zone flow and Boussinesq's equation for groundwater flow.

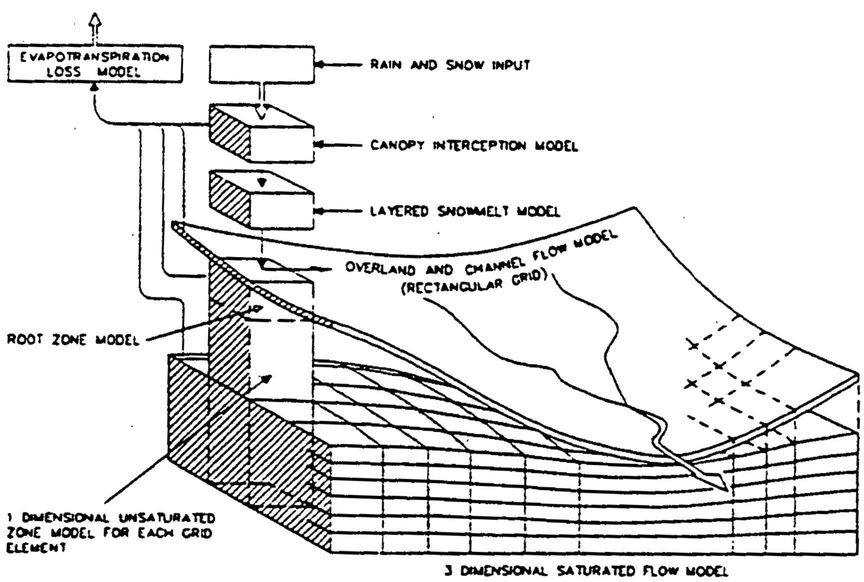

Figure 6. Schematic diagram of a catchment and the MIKE SHE quasi three-dimensional distributed physically-based model (DHI, 1993).

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Distributed physically-based models have been used for a couple of decades on a routine basis for the simulation of single hydrological processes, so that almost all groundwater models and most unsaturated zone flow models belong to this type. A first attempt to outline the potentials and some of the elements in a distributed physically-based model on a catchment scale was made by Freeze and Harlan (1969). Today, several general purpose catchment model codes of this type exist such as SHE (Abbott et al., 1986), MIKE SHE (Refsgaard and Storm, 1995), IHDM (Beven et al., 1987) and THALES (Grayson et al., 1992).

Distributed physically-based models give a detailed and potentially more correct description of the hydrological processes in the catchment than do the other model types. Moreover, they are able to exploit the quasi-totality of all information and all knowledge that is available concerning the catchment that is being modelled. The distributed physically based models can in principle be applied to almost any kind of hydrological problem. However, in practice, they will be used complementary to the other model types for cases where the other models are not suitable. Some examples of typical applications are:

* Prediction of the effects of catchment changes due to human interference in the hydrological cycle, such as changes in land use (including urbanization), groundwater development and irrigation. Since parameters of the model tend to be more physically based, the change in parameter values corresponding to the catchment changes can some times be estimated directly.
* Prediction of runoff from ungauged catchments and from catchments with relatively short records. As opposed to the lumped conceptual models, which require long historical time series of rainfall, runoff and evaporation data for the assessment of parameters, the parameters of the distributed physically-based models may be assessed directly from intensive, short-term field investigations.
* Water quality and soil erosion modelling for which a more detailed and physically correct simulation of water flows is important.

However, even with the most advanced of the distributed, physically based models, significant deficiencies still persist at the level of the process descriptions. Hence, the term 'white box', suggesting as it does that everything within the model is transparent and correct, is not fully justified, and 'light grey' might be more appropriate.

Due to the significant progress being made within the scientific community in the direction of an enhanced physical understanding, this model type will surely continue to be improved over the next many years.

3.3 STOCHASTIC TIME SERIES MODELS

A stochastic time series model treats the sequence of events that it comprehends as time-dependent. Traditionally, a stochastic model is derived from a time series analysis of the historical record. The stochastic model can then be used for the generation of long hypothetical sequences of events with the same statistical properties as the historical record. The technique of generating several synthetic series with identical statistical properties is denoted the *Monte Carlo technique*. These generated sequences of data can then be used in the analyses of design variables and their uncertainties, such

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as these may arise, for example, when estimating reservoir storage requirements. An overview of stochastic time series modelling systems is given in Salas (1992).

With regard to process description, the classical stochastic simulation models are comparable to the empirical, black box models described in Subsection 3.2.1 above. Hence the stochastic time series models are in reality composed of a simple deterministic core (the black box model) contained within a comprehensive stochastic methodology.

3.4 JOINT STOCHASTIC-DETERMINISTIC MODELS

On one hand, a substantial part of the hydrological processes, including the spatial and temporal variations of hydrological parameters and variables, can today be described using deterministic simulation models. On the other hand, the available information on parameter values and input variables will always be incomplete. This lack of knowledge is an important source of uncertainty in hydrological simulation.

Acknowledging this duality, several model types based on a joint stochastic-deterministic approach have been developed. These models are composed of two, in principle equally important parts, namely a deterministic core within a stochastic frame. In contrast to the stochastic simulation models described in Section 3.3 above, the deterministic core is here composed of more comprehensive models, whether of the lumped conceptual or the distributed physically-based type.

In the following, some examples illustrating different types of joint deterministic-stochastic models are given.

3.4.1 State space formulations - Kalman filtering

The state space theory and the Kalman filtering technique (e.g. Gelb, 1974) are powerful mathematical tools originally developed within the field of statistical control theory for linear systems, but later extended to comprehend non-linear systems also. They are now being applied increasingly also in hydrology.

The key model variables are recognized as stochastic variables that are parameterized in terms of their mean and standard deviations. The input variables (e.g. rainfall data) are similarly described by a mean value (the recorded value) and a standard deviation (the uncertainty). In this way it is possible to calculate how uncertainty on, for example, input data propagates through the model and causes uncertainties on model state variables and output results.

Combinations of Kalman filtering and lumped, conceptual rainfall-runoff models have been realised, amongst others for the Sacramento modelling system (Kitanidis and Bras, 1978; Georgakakos et al., 1988) and for the NAM system (Refsgaard et al., 1983; Storm et al., 1988).

3.4.2 Spatial variabilities of parameter values and stochastic PDE's

Subsurface hydraulic parameters exhibit very large spatial variabilities even on very small scales. Therefore, it is not realistic to give a correct complete deterministic description in all details. In most cases, however, knowledge of the detailed flow pattern is of little interest; it is rather the impact of the spatial heterogeneity on the

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overall flow pattern which is important.

One way of treating this problem is to represent (some of) the hydraulic parameters as realizations of stochastic variables with known statistical properties (probability distribution type, mean, standard deviation, spatial autocorrelation). As a result of following this approach, the governing equations become what are often described as stochastic partial differential equations (PDEs), which are of course usually much more difficult to solve than are traditional deterministic PDEs. Two possible approaches to the stochastic PDE's are:

* The Monte Carlo technique, whereby the deterministic model is run several times using different (equally probable) realizations of the parameter field. Finally, statistical analyses are made of the results from all the deterministic simulation runs. The advantage of this approach is that the deterministic model can be preserved, whereas the main disadvantage is the very large CPU requirement. Classical examples of this approach are provided by Smith and Freeze (1979a,b) and Freeze (1980) for groundwater flow and rainfall-runoff processes, respectively. Zhang et al. (1993) demonstrated a methodology of using the Monte Carlo technique to determine the effect of uncertainty in model parameters and rainfall on the uncertainty of model responses and the further impact of these uncertainties on sample size requirements for simulating solute transport through soils.
* The stochastic PDE is simplified and solved analytically. This is often carried out by elegant mathematical solution techniques and has proven very useful for obtaining general insights into fundamental research problems. However, this methodology puts so significant limitations on the range of data input, the size of the problem, the boundary conditions and other aspects that it appears to have had limited practical application. This methodology has been used especially within the fields of unsaturated zone and groundwater flow and transport for research purposes, see, for example, Gelhar (1986) and Dagan (1986). Jensen and Mantoglou (1992) applied a stochastic theory to modelling of a small experimental field. This theory was subsequently incorporated into MIKE SHE and applied to catchment size problems by Sonnenborg et al. (1994).

The joint stochastic-deterministic methods may be seen as extensions of the deterministic simulation models described in Section 3.2. The output result of a deterministic model simulation is in principle one time series for each of the model variables. The joint stochastic-deterministic models, on the other hand, produce uncertainty bands (such as are described by mean values and standard deviations) for each of the predicted time series. Thus, the joint stochastic-deterministic models are able to transfer the inherent uncertainty on the input variables or effects of non-described variability of spatial parameter values to probabilistic descriptions of the output variables.

A more comprehensive approach also considering the uncertainty in the model structure and process equations is the GLUE technique introduced by Beven and Binley (1992).

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4. Modelling of Spatial Variability of Hydrological Parameters

Hydrological parameter values exhibit very large spatial variations in nature. Thus, detailed field measurements indicate that, for instance, the hydraulic conductivity may vary by several orders of magnitude within a small field which is traditionally characterised as 'homogeneous' and belonging to the same soil type (Nielsen et al., 1973). One of the key differences between lumped and distributed models are the fundamentally different ways in which they take the spatial variability into account.

Consider for illustrational purposes how the Stanford and the MIKE SHE, as typical representatives for the lumped conceptual and the distributed physically-based model types, describe the infiltration process in the unsaturated zone over a catchment.

In the Stanford an assumption of spatial variability of the infiltration capacity is in a simplified way built into the infiltration equation of the model code. Thus, the spatial variability of the key soil parameter is implicitly taken into account. As a result the mechanism for generation of overland flow functions as a contributing area approach.

In MIKE SHE the infiltration calculations are based on Richards' equation which is assumed to be valid theoretically for single-domain porous media flow. Being a distributed model the catchment is divided into a number of grids, and the infiltration equation is thus used on a number of unsaturated zone columns, each of which is characterised by soil hydraulic parameters. Thus, in a distributed model the spatial variability is taken into account explicitly through the variability of the model parameter values. In principle, the model parameters are different in the different soil columns. However, in practice the parametrization procedure usually prescribes identical parameter values for all grid points with the same soil type, and furthermore there will always be significant variability within a grid which cannot be accounted for by a deterministic approach. Thus, unless the variability among soil groups is significantly larger than the variability within soil groups (and this is often not the case) MIKE SHE does not explicitly take all the spatial variability into account. The result is that the overland flow generation in the catchment theoretically becomes too much of an 'on/off' process. In practice, this effect is often overshadowed by other dominating processes or it can be compensated through calibration.

Hence, even in fully physically-based models the spatial variability of hydrological parameters will most often not be at all fully described. In order to take the variability more fully into account, a joint stochastic-deterministic model as outlined in Subsection 3.4.2 may be adopted.

Another approach is adopted in TOPMODEL (Beven, 1986), where the spatial variability of soil properties is built into the process equations. TOPMODEL does not fit into one of the schematic model classes defined in Section 3, but may be characterized as a semi-distributed semi-physically-based modelling system. The advantages of TOPMODEL as compared to traditional lumped conceptual systems are the explicit accounting of the spatial variability and the direct use of spatial data such as topography and channel system together with semi-distributed calculations of hydrological variables.

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5. Classification according to Technological Level

A practice of modelling can be categorized according to Abbott et al. (1991) according to its technological level in different generations of modelling systems:

First Generation - Computerised Formulae. The first generation dates back to the early 1950s. The introduced computer codes mainly aimed at making the methods of numerical calculations that had been developed for human computation easier and quicker in through the application of the very simplest numerical methods. At one time it was popular saying that this was to use the computer as a 'super slide rule' but nowadays, of course, few persons know anymore what even a slide rule was.

Second Generation - One-Off Numerical Models. The second generation of modelling appeared in the 1960s. It was involved in construction, applying and developing models for the most part by university or research institutes, for solving one specific problem. The practice of the second generation of modelling was generally restricted to the group of persons who had developed the code for one particular geographical domain, but the models so developed were not generally applicable to use by other persons or to similar problems under other external conditions. Second generation models generally required comprehensive user experience within the fields of computers (hardware and software) and the related numerical techniques.

Third Generation - Generalised Numerical Modelling Systems. Third generation modelling was directed to constructing generalised modelling systems with which many different problems could be solved by the use of one and the same computer code. Third generation modelling systems emerged in the 1970s in hydrodynamics. In hydrology the first distributed physically-based systems were made in the beginning of the 1980s, while the much simpler lumped conceptual systems appeared in the 1960s. These systems were mainly applied by computer-experienced specialists. Some basic knowledge of numerical methods was also required by the user. Traditionally, third generation systems have been based on main frame computers, but most of them can run on PC's today. However, this approach is now largely superceded by fourth generation modelling.

Fourth Generation - The Industrial User-friendly Software Product: These are user-friendly software products that can be applied by engineering and scientific professionals. Fourth generation systems differ from third generation systems in the following ways:

* They are usually PC (or workstation) based.
* They are fully menu based, providing interactive execution with on-line help menus.
* They provide comprehensive error messages and automatic checks for obviously erroneous input data.
* They provide more powerful graphics facilities both on monitor, printer and plotter devices.
* They are generally much better documented and more well-proven due to a larger installation base.
* They are more easily transferable and installable.
* They have a much wider circulation (being distributed in one or two orders of magnitude more copies).

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These systems therefore make little or no demands on user experience in computer systems or numerical techniques, but they still require user experience in modelling. The first fourth generation systems appeared in the mid 1980s.

Fifth Generation - 'Intelligent' Modelling System. These begin with modelling systems designed for technically-skilled but non-expert users. They include numerical stability monitoring and diagnosis tools, interfaces to standard CAD and data base systems, and knowledge base system frames applicable in decision making. They merge into hydroinformatics tools generally, such as diagnostic and real-time control systems and management support systems (e.g. Abbott, 1991; IAHR, 1994; Verwey et al, 1994).

At present only a few fourth generation systems are operational and fifth generation systems are still mainly at an experimental stage outside of real-time control applications for urban drainage systems (e.g. Gustafsson et al., 1993). By way of an example the development history of DHI's river modelling system, MIKE 11, is illustrated in Fig 7. In order to achieve a large dissemination of modelling technology to a considerable part of the professional community (and not only to modelling experts) experience shows that fourth generation systems are required. Furthermore, it is believed that fifth generation systems extending to hydroinformatics systems are required in order for the modelling technologies to achieve their full potential in terms of practical application.

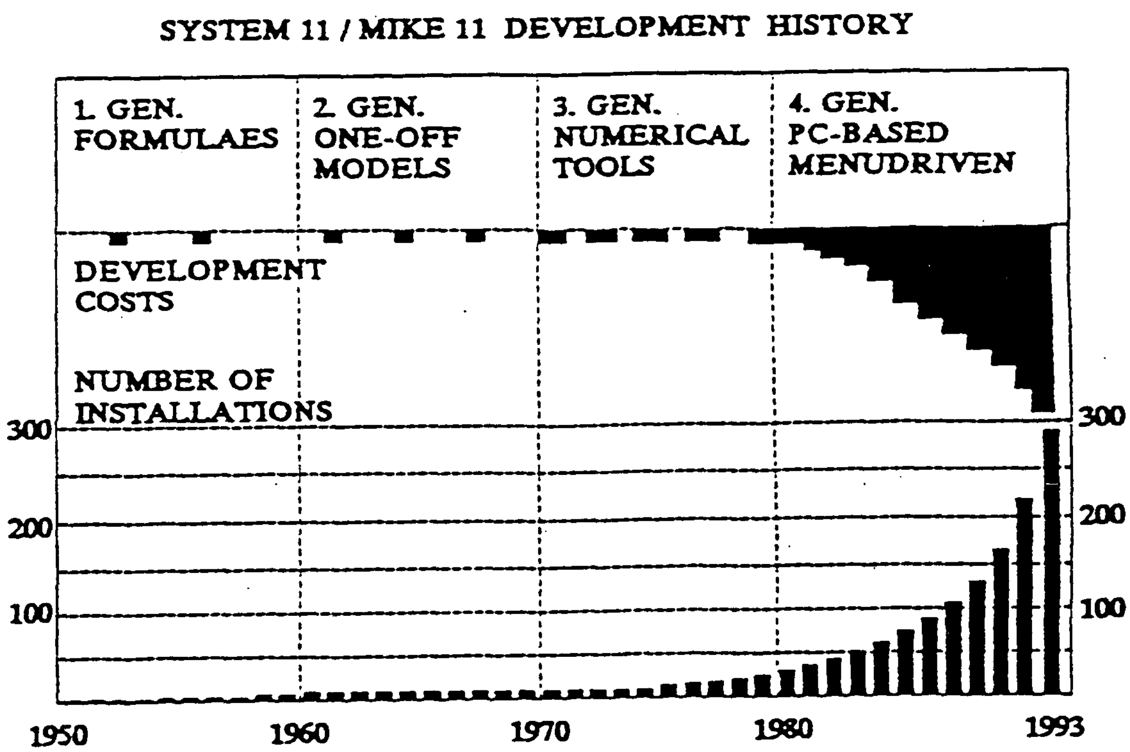

Figure 7. The development history of DHI's river modelling system MIKE11.

6. Acknowledgement

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CHAPTER 3
CONSTRUCTION, CALIBRATION AND VALIDATION OF HYDROLOGICAL MODELS

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1. Introduction

In Chapter 2 of this book a terminology was defined and the different steps in a modelling application was put into the framework of a modelling protocol. In the present chapter some of the elements in such protocol are addressed in more details with special reference to distributed models.

The different types of error sources in hydrological simulation are outlined in Section 2. Section 3 gives an introduction to goodness of fit and accuracy criteria. Section 4 deals with the procedure of model construction including parametrisation. Section 5 provides an overview of different procedures adopted for model calibration, while the model validation issue is dealt with in Section 6. Finally, the issue of credibility or degree of validity of a generic modelling system is addressed in Section 7.

2. Reasons of Uncertainty in Hydrological Modelling

2.1 DETERMINISTIC MODELLING CONCEPT

The concept of deterministic mathematical modelling can be illustrated as in Fig. 1, where the physical system, in this case a catchment, is shown to the left. The mathematical model, which is the sign-representation of the physical system is shown to the right. Both the quantification of the physical system through monitoring and the representation through mathematical modelling are subject to errors.

The temporal and spatial variability of inflows, outflows and internal conditions are quantified from measurements at discrete locations. Unfortunately these measurements alone can neither provide us with any complete picture of the conditions inside the catchment area nor an understanding of the mass, energy and other exchanges between the catchment and its surroundings. Firstly, because it is not feasible to measure all the spatial and temporal variations of flows and state variables, and secondly because the measurements include errors. Consequently, the model will be using an approximate dataset which to some degree contain sampling errors. On the other hand, by definition, the physical system experiences the real data and responds to this real input. However, the response is measured with some uncertainty.

When the hydrological model is used to simulate the behaviour of the physical

M. B. Abbott and J. C. Refsgaard (eds.), Distributed Hydrological Modelling, 41–54.

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system, it produces output data, which is affected by the approximate input data. In order to test the accuracy of the developed model simulated output is then compared with the recorded data which also is subject to uncertainties. A desired model accuracy is achieved by changing the parameter values used in the model until a satisfactory agreement between simulated and the recorded variables is obtained. This parameter adjustment process is the process of calibration.

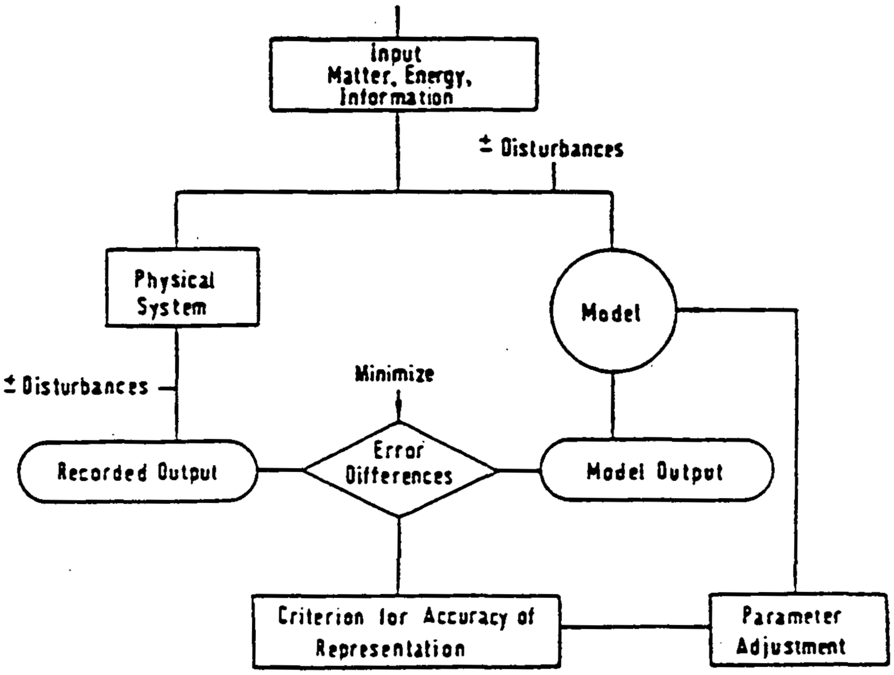

Figure 1. The concept of deterministic mathematical modelling. After Fleming (1975).

2.2 SOURCES OF UNCERTAINTY

Differences between recorded data and simulated model output arise basically from four sources of uncertainty:

(1) Random or systematic errors in the input data, i.e. precipitation, temperature and evapotranspiration etc. used to represent the input conditions in time and space over the catchment.

(2) Random or systematic errors in the recorded data, i.e. the river water levels, groundwater heads, discharge data or other data used for comparison with the simulated output.

(3) Errors due to non-optimal parameter values.

(4) Errors due to an incomplete or biased model structure.

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Although a disagreement between simulated and recorded data is the combined effect of all four error sources, only error source (3) can be minimized during the calibration process. The measurement errors, error sources (1) and (2), are the "background noise" which determine the maximum achievable agreement, which the modeller can hope to obtain. Changes in parameter values or model structure cannot and should not improve the results beyond that. The objective of a calibration process is therefore to reduce the error source (3) until it becomes insignificant compared to the data error sources (1) and (2).

During a calibration process it is of utmost importance to ensure that a clear distinction is drawn between the different error sources, so no attempt is made to compensate for errors from one source by adjustments within another source, such as compensating for a data error by parameter adjustments. Otherwise the calibration will approach curve fitting, which may result in a reasonable fit within the calibration period but may incidently provide unreliable predictions.

It is also important that the modeller recognises the limitations of a chosen modelling approach and is not trying to impose a certain conceptual representation of the physical system because it fits to the mathematical formulation in the model code applied.

3. Goodness of fit and accuracy criteria

During the calibration procedure different accuracy criteria can be used to compare the simulated and measured data. This allows us to define an objective measure of the goodness of fit associated with each set of model parameters and estimate the parameter values which provide the best overall agreement between model output and measured data. However, the selection of an appropriate criterion is complicated by the variation in the sources of error discussed in the previous section. It further depends on the objective of the model simulation (e.g. to simulate flood peaks or low flows) and on the type of information available to check model output variables, such as phreatic surface levels, soil moisture contents, stream discharges or stream water levels.

No single criterion is entirely suitable for all variables, and even for a single variable it is not always easy to establish a satisfactory criterion. Hence a large number of different criteria has been developed. Green and Stephenson (1986) discuss the problem in detail and list 21 approaches which can be used for single-event simulations depending on the simulation objectives, range of simulation conditions and other factors. Criteria for the detection of systematic errors relevant to long-term simulations are discussed by Aitken (1973).

It is possible to calibrate a model by optimizing just one criteria. However, a calibration based on a 'blind' optimization of a single numerical criterion can produce physically unrealistic parameter values which, if applied to a different time period, may give poor simulation results. Green and Stephenson (1986) conclude that no single criterion is sufficient to assess adequately the overall measure of fit between a computed and an observed hydrograph, particularly in view of the many objectives behind hydrological modelling. At the same time, it should be remembered that the criteria measure only the correctness of the estimates of the hydrological variables generated

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by the model and not the hydrological soundness of the model relative to the processes being simulated. It is therefore recommended that, in a calibration, numerical criteria be used for guidance only.

Finally, the benefit of using graphical comparison of e.g. simulated and observed hydrographs should be emphasized. Although the analysis becomes more subjective, graphical comparisons provide a good overall indication of the models capabilities, they are more easily assimilated and may impart more practical information than statistical functions alone. Fig. 2 illustrates the combined use of numerical and graphical comparisons of runoff data. In the figure the flow duration curve error index, EI, is a numerical measure of the difference between the flow duration curves of the simulated and observed daily flows (perfect agreement for EI = 1). Similarly, R2 refers to the Nash-Sutcliffe coefficient (Nash and Sutcliffe, 1970), computed on the basis of the sequence of observed and simulated monthly flows.

4. Model construction

Model construction is the process of preparing the data in a correct format and entering them into a set of input data files required by the model code so that the first model run can be made as the first step in the subsequent model calibration.

For lumped conceptual model codes or similar type of model codes which only requires few parameter values for a set of empirical equations, the model parameters can generally not directly be estimated from catchment characteristics, but must be estimated through calibration. The data requirements are therefore very small and the data preparation is usually confined to construction of data files containing time series of climatic input data and a corresponding time series of control data e.g. river discharges. These can usually be prepared very quickly.

For distributed physically-based models, the data preparation phase is more complicated and involves often a comprehensive programme of work. A fully distributed physically-based model contains only parameters which can be assessed from field data, so that, in principle, a calibration would not be necessary if sufficient data were available. In practice, distributed physically-based model codes are most often applied at scales on which the parameter values cannot be directly assessed, and will therefore require calibration. In the subsequent calibration phase, allowed parameter variations may be restricted to relatively narrow intervals compared to parameter values related to the empirical functions in empirical or lumped conceptual models.

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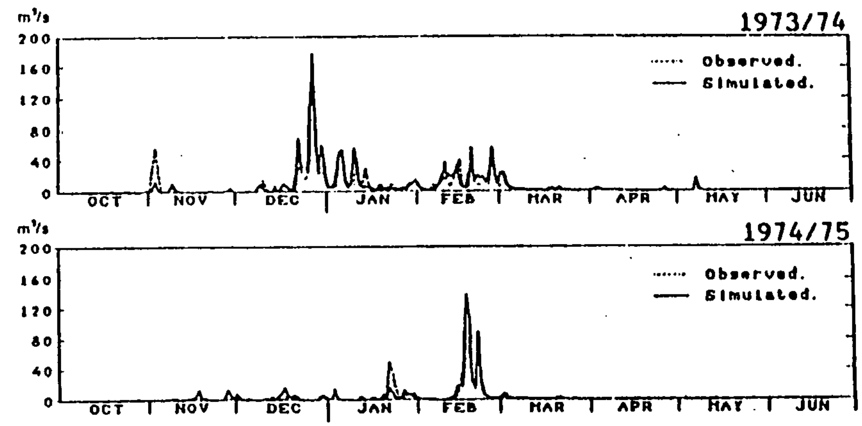
Annual Water Balance

YEAR	RAIN	Epot	Qobs	Qsim	DFF
1971/72	916.	1808.	89.	118.	27.
1972/73	379.	2201.	2.	23.	21.
1973/74	1385	1450	440	558	98
1974/75	1043.	1789.	217.	262.	45.
1975/76	897	1784.	89.	102.	13.
Average	915	1846	171	212	41



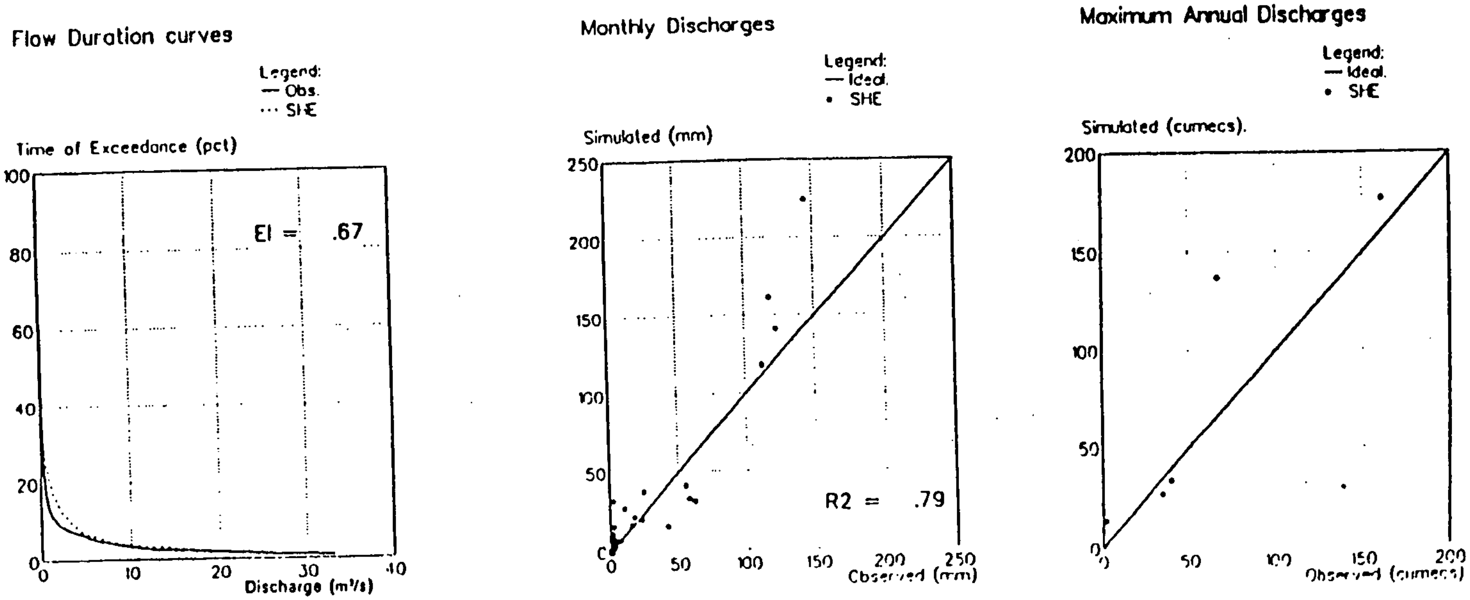


Figure 2. Example of combined use of numerical and graphical performance criteria. Example from proxy-basin validation test of MIKE SHE to the Lundi catchment in Zimbabwe (DHI, 1993).

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Distributed models describe the spatial variations in catchment characteristics and input data in a network of grids points. In order to provide a sufficient detail the spatial resolution often requires several thousands of grid points, each of which is characterized by one or more parameters. Although the parameter values in principle (as in nature) vary from grid point to grid point, it is neither feasible nor desirable to allow the parameter values to vary so freely. Instead, a given parameter should only reflect the significant and systematic variation described in the available field data. as exemplified by the practice of using representative parameter values for individual soil types. vegetation types or geological layers. This process of defining the spatial pattern of parameter values, which is denoted *parametrisation*, effectively reduces the number of free parameter coefficients which needs to be adjusted in the subsequent calibration procedure.

An example of a parametrisation procedure used for constructing SHE models for catchments in India is presented in Refsgaard et al. (1992) and Jain et al. (1992). As an example the 820 km^2 Kolar catchment is parameterised into three soil classes and 10 land use/soil depth classes. For the soil type classes calibration was allowed for the hydraulic conductivity in the unsaturated zone (for each soil type class the conductivity could vary among three different land uses => nine parameter values). For the land use/soil depth classes the calibration parameters comprised soil depths (10 parameters in total) and the Strictler overland flow coefficients for four land use types (four parameters in total). Further three parameters were subject to calibration (hydraulic conductivity in the saturated zone, an (empirical) by-pass coefficient and a surface retention parameter; all kept constant throughout the catchment). Although the 26 calibration parameters could not be assessed from field data alone, but had to be modified through calibration, the physical realism of the parameter values resulting from the subsequent calibration procedure could be evaluated from available field data.

An example of parametrisation procedure for a combined groundwater/surface water model for an 800 km^2 area near Aarhus. Denmark is given by Refsgaard et al. (1994).

A rigorous parametrisation procedure is crucial in order to avoid methodological problems at the subsequent phases of model calibration and validation. The following points are important to consider in the parametrisation procedure:

* The parameter classes (soil types. vegetation types, climatological zones, geological layers, etc.) should be selected so that it becomes easy, in an objective way, to associate parameter values. Thus the parameter values in the different classes should to the highest possible degree be assessable from available field data.
* It should explicitly be evaluated which parameters can be assessed from field data alone and which need some kind of calibration. For the parameters subject to calibration physically acceptable intervals for the parameter values should be estimated.
* The number of real calibration parameters should be kept low, both for practical and methodological points of view. This can be done, for instance, by fixing a spatial pattern of a parameter but allowing its absolute value to be modified through calibration.

5. Calibration methods

In a calibration procedure, values are estimated for those parameters which cannot be assessed directly from field data. In principle, three different calibration methods can be applied:

(1) Trial-and-error, manual parameter adjustment.

(2) Automatic, numerical parameter optimization.

(3) A combination of (1) and (2).

In Subsection 5.1 the three methods as well as their advantages and disadvantages are described. The specific status on calibration techniques within the areas of lumped conceptual and distributed physically-based modelling are summarized in Subsections 5.2 and 5.3.

5.1. ADVANTAGES AND DISADVANTAGES OF USING TRIAL-AND-ERROR AND AUTOMATIC METHODS.

5.1.1 Trial-and-error manual calibration.

The trial-and-error method implies a manual parameter assessment through a number of simulation runs. This method is by far the most widely used and is the most recommended method, especially for the more complicated models. A good graphical representation of the simulation results is a prerequisite for the trial-and-error method.

5.1.2 Automatic parameter optimization.

Automatic parameter optimization involves the use of a numerical algorithm which finds the extremum of a given numerical objective function. The purpose of automatic parameter optimization is to search through as many combinations and permutations of parameter levels as possible to achieve the set which is the optimum or 'best' in terms of satisfying the criterion of accuracy.

The *advantages of automatic parameter optimization* over the trial-and-error method are:

* Automatic optimization is fast, because almost all work is carried out by the computer.
* Automatic optimization is less subjective than the trial and error method, which to a large degree depends on visual hydrograph inspection and the personal judgement of the hydrologist.

The *disadvantages of automatic parameter optimization* include:

* The criterion to be optimized has to be a single numerical criterion based on a single variable. As discussed in Section 3, though, the selection of an appropriate criterion under these constraints is a complicated and, in its turn, often quite subjective task.
* If the model contains more than a very few parameters, the optimization will probably result in the location of a local optimum instead of the global one.
* Most theories behind the search algorithms assume that the model parameters are mutually independent. This assumption is usually not justified.
* An automatic routine cannot distinguish between the different error sources mentioned in Section 2. Accordingly, an automatic optimisation algorithm may try

to compensate, for example, for data errors by parameter adjustments, with the results that the parameter values often become physically unrealistic and give poor simulation results when applied to a period different from the calibration period.

To these purely technical considerations may be added a more general one (Todini, 1988). To calibrate the parameters of a model by the optimization of an objective function means adopting some statistical technique (least squares, linear or non-linear regression, maximum likelihood, etc) based on an analysis of residuals, while completely neglecting the physical characteristics of the model. In other words, a procedure of automatic calibration, rather than capitalising on prior knowledge intrinsic to the structure of the model, avoids it, and thus emphasises the uncertainty inherent in every statistical analysis.

5.1.3 Combination of trial-and-error and automatic parameter optimization.
A combination could involve, for example, an initial adjustment of parameter values by trial-and-error to delineate rough orders of magnitude, followed by a fine adjustment using automatic optimization within the delineated range of physically realistic values. The reverse procedure is also possible: first carrying out sensitivity tests by automatic optimization to identify the important parameters and then calibrating them by trial-and-error. The combined method can be very useful but does not yet appear to have been widely adopted in practice.

5.2. CALIBRATION OF LUMPED CONCEPTUAL MODELS

The most widely used method for the calibration of conceptual rainfall-runoff models is still the manual trial-and-error method. These models are simple and fast to operate. However their calibration requires an experienced user. An experienced hydrologist can usually achieve a calibration using visual hydrograph inspection within 5-15 trial runs.

Much research work during the past two decades has concentrated on establishing appropriate automatic parameter estimation methods based on traditional numerical search algorithms. One of the oldest methods, which was very popular in the 1970s, is Rosenbrock's method (Rosenbrock, 1960). A comparison of different search optimization algorithms and a discussion of some problems relating to automatic calibration are reported in Gupta and Sorooshian (1985) and Sorooshian et al. (1993).

A different method, which has emerged within the last few years, is based on expert system technology. An expert system can be defined as a computer program in which knowledge is introduced within a first-order predicate logical frame that allows it to operate at an expert level. The idea in this approach is to let an experienced user teach the computer how he or she, in a trial-and-error process, decides which parameter adjustments to carry out before the next run. One example of a rule-based expert system is given by Azevedo et al (1993).

Automatic parameter optimization was extensively attempted in practice during the 1970s, but is not used very much today owing to the problems outlined in Subsection 5.1.

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5.3. CALIBRATION OF DISTRIBUTED PHYSICALLY-BASED MODELS

Comprehensive research has been carried during the past decade on establishing and testing methods for automatic parameter optimization methods in groundwater modelling (described as inverse modelling), see e.g. Keidser and Rosbjerg (1991). Computer programmes comprising inverse methods applicable for two-dimensional flow problems are available but are not much used in practice. For solute transport problems and for three-dimensional models the inverse methods are still at the experimental stage.

For distributed hydrological catchment modelling systems such as MIKE SHE, which simulates many catchment processes and the dynamic interactions between them, suitable optimization methods are not yet available, and indeed are not even at a research stage. There are many problems related to the application of automatic parameter optimization methods in such complex models. One important problem relates to the necessity of having a multicriteria objective function, containing, for example, references to discharge, soil moisture, groundwater levels and possibly also concentration values at several points within the catchment.

However, the massive investments in inverse methods in other fields, such as meteorology and oceanography may be expected to provide tools that will be applicable to hydrology also. Certainly, inverse methods currently appear as the most promising candidates for promoting the practice of distributed hydrological models over the next decade.

6. Model validation

6.1. INTRODUCTION

If a model contains a large number of parameters it is often possible to find a combination of parameter values which provides an acceptable match between measured data and simulated output data for a short simulation period. This may be true even if the structure of the model is inappropriate or the conceptual understanding of the hydrological system is incorrect. A good match does not necessarily guarantee that a correct set of parameter values has been found, because the calibration may have been achieved purely by numerical curve fitting without considering whether the parameter values obtained are physically reasonable. This may also be illustrated by the fact that if identical models were calibrated by different people apparently equally good calibrations would be achieved based on different combinations of parameter values.

In order to assess whether a calibrated model can be considered valid for subsequent use it must be tested (validated) against data different from those used for the calibration (e.g. Stephenson and Freeze, 1974). Klemes (1986) states that a simulation model should be tested to show how well it can perform the kind of task for which it is specifically intended.

According to the methodology established in Chapter 2 of this book model validation implies substantiating that a site specific model can produce simulation results within the range of accuracy specified in the performance criteria for the particular study. For this purpose, a general scheme for validation tests is outlined in Subsection 6.2, while a discussion of different validation requirements for lumped and distributed models is given in Subsection 6.3.

6.2. SCHEME FOR CONSTRUCTING SYSTEMATIC VALIDATION TESTS

Klemes (1986) has proposed a hierarchical scheme for systematising validation tests of hydrological simulation models. The scheme is said to be hierarchial because the modelling tasks are ordered according to their increasing complexity and the demands of the testing increase in the same direction. The scheme was originally developed for rainfall-runoff modelling, but the methodology can be applied more generally.

Klemes (1986) distinguished between simulations conducted for the same station (catchment) as was used for calibration and simulations conducted for ungauged catchments. He also distinguished between cases where catchment conditions (climate, land use, ground water abstraction etc.) are stationary and cases where they are not.

This division leads to the definition of four basic categories of typical modelling tests:

A: *Split-sample test:* Calibration of model based on, say, 3-5 years of data, and validation on another period of similar length.

B: *Differential split-sample test:* Calibration of a model based on data before catchment change occurred, the adjustment of model-parameters to characterise the change and a validation based on the subsequent period.

C: *Proxy-basin test:* No direct calibrations are allowed but advantage is taken of information available from other gauged catchments. Hence, validation will comprise the identification of a gauged catchment deemed to be of a similar nature to the validation catchment, the initial calibration, the transfer of the model, including the adjustment of parameters to reflect actual conditions within the validation catchment, and validation.

D: *Proxy-basin differential split-sample test*: Again no direct calibration is allowed but information from other catchments may be used. Hence, validation will comprise an initial calibration on other relevant catchment, the transfer of the model to the validation catchment, the selection of two parameter sets to represent the periods before and after the change, and subsequent validations on both periods.

6.2.1. Split-sample test

The split-sample test is the classical test, being applicable to cases where there is sufficient long time series of control data for both calibration and validation, and where the catchment conditions remain unchanged, i.e. are stationary.

The available data record is divided into two parts. A calibration is carried out in turn for each part and then validated against the other part. Both these calibration and validation exercises should give acceptable results.

The main problem associated with the split-sample test is that not all the available data are used for the calibration. Therefore the data record should be of such length that, when split into parts, these parts can support an adequate calibration. On the other

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hand, if both split-sample tests produce acceptable results, a final calibration of the model can make use of the full record.

6.2.2. Proxy-basin test

This test should be applied when there is not sufficient data for a calibration of the catchment in question. If, for example, streamflow has to be predicted in an ungauged catchment Z, two gauged catchments X and Y within the region should be selected. The model should be calibrated on catchment X and validated on catchment Y and vice versa. Only if the two validation results are acceptable and similar can the model command a basic level of credibility with regard to its ability to simulate the streamflow in catchment Z adequately.

6.2.3. Differential split-sample test

This test should be applied whenever a model is to be used to simulate flows, groundwater levels and other variables in a given gauged catchment under conditions different from those corresponding to the available data. The test may have several variants depending on the specific nature of the modelling study.

If for example a simulation of the effects of a change in climate is intended, the test should have the following form. Two periods with different values of the climate parameters of interest should be identified in the historical record, such as one with a high average precipitation, and the other with a low average precipitation. If the model is intended to simulate streamflow for a wet climate scenario, then it should be calibrated on a dry segment of the historical record and validated on a wet segment.

Similar test variants can be defined for the prediction of changes in land use, effects of groundwater abstraction and other such changes. In general, the model should demonstrate an ability to perform through the required transition regime.

6.2.4. Proxy-basin differential split-sample test

This is the strongest test to pass for a hydrological model, because it deals with cases, where there is no data available for calibration, and where the model is intended for prediction of non-stationary conditions.

6.3. DIFFERENT VALIDATION REQUIREMENTS FOR LUMPED AND DISTRIBUTED MODELS

The validation procedure is basically the same for lumped and distributed model codes, but because of the differences in model structures, modes of operation and objectives of application, the validation requirements are much more comprehensive for distributed models. Traditional validation based on comparing simulated with observed outflows at the catchment outlet still remains the only option in many practical cases. However, as emphasized by Rosso (1994) this method is poorly consistent with spatial distributed modelling. The differences are summarized in Table 1, which clearly illustrates the need for multicriteria, multi-scale validation criteria.

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TABLE 1. An illustration of the need for the incorporation of multicriteria and multi-scale aspects in methodologies for the validation of distributed models.

	LUMPED CONCEPTUAL	DISTRIBUTED PHYSICALLY-BASED
Output	At one point: • Runoff => *single variable*	At many points: • Runoff • Surface water level • Ground water head • Soil moisture => *multi variable*
Success criteria (excl problem of selecting which statistical criteria to use)	Measured <=> simulated * Runoff, one site => ***single criteria***	Measured <=> simulated • Runoff, multi sites • Water levels, multi sites • Groundwater heads, multi sites • Soil moisture, multi sites => ***multi criteria***
Typical model application	Rainfall-runoff * stationary conditions • calibration data exist	Rainfall-runoff, unsaturated zone, groundwater, basis for subsequent water quality modelling Impacts of man's activity * non-stationary conditions some times • calibration data do not always exist
Validation test	Usually "Split-sample test" is sufficient => ***well defined practise exist***	More advanced tests required: • Differential split sample test • Proxy basin test => ***need for rigorous methodology***
Modelling scale	Model: catchment scale Field data: catchment scale => ***single scale***	Model: depends on discretization Field data: many different scales => ***multi scale*** problems

The table shows that because the project objectives and the output requirements are more demanding for distributed models, the success criteria has to be more rigorous. Validation against one single discharge station (e.g. at the catchment outlet) is sufficient if the purpose is to generate streamflow values at that location, but not sufficient to draw any conclusions about the internal representation of the flow conditions within the catchment. Based on the single discharge station the total water balance may be correct, but for example a systematic underestimation of the actual evapotranspiration could lead to unrealistic trends in the groundwater heads over the entire or parts of the catchment.

Basically, a success criterion needs to be fulfilled for each output variable which we intend to make predictions for. Multi-site calibration/validation is needed if spatially distributed predictions are required, and multi-variable checks are required if predictions of the behaviour of individual sub-systems within the catchments are needed.

Styczen and Storm (1995) showed that calibration of both streamflow and the trend in groundwater heads was required to ensure that the simulated groundwater recharge rates for estimation of nitrate leakage was correct.

In Storm and Punthakey (1995) MIKE SHE was used to simulated the groundwater table variations in an irrigation district. Data from a very detailed groundwater observation network was available for the calibration, but no drainage flow data was available. Although the model was used to estimate groundwater accessions from the land surface, they could not guarantee that the actual flow passing through the groundwater system was simulated correctly.

7. The Credibility of a Generic Modelling System

According to the terminology defined in Chapter 2 of this book a modelling system can, in principle, never be validated. Instead of a full validation we can think of the degree of validity as the *credibility of a given modelling system*. The degree of validity of a modelling system is expressed in the first and most immediate place by the sum of all successful validations of all models that have been constructed and operated to date using the modelling system. As the number of such successful model increases, so the credibility of the system itself grows in strength.

Behind this, most superficial of views, lies the assumption that the modelling system is in fact being improved on the basis of the operating experience; that it is functioning within its market, tracking the needs of that market and thus learning from this market. From this point of view, the development of a modelling system is not one that leads directly to a finished, rounded and complete product, but it is rather a *process of adaption through evolution*. Thus, although the modelling system is indirectly a product, it is one that is constantly evolving, so that its evolution corresponds to a process. The general principles of such a development have been expressed by Floyd (1987).

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CHAPTER 4

DISTRIBUTED PHYSICALLY-BASED MODELLING OF THE ENTIRE LAND PHASE OF THE HYDROLOGICAL CYCLE

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1. Introduction

Physically-based distributed hydrological model codes have been developed from a need to analyze and solve specific hydrological problems often required in multi-objective and multi-decision management investigations. These problems may differ in type and scale, but have usually one thing in common, namely that in order to obtain a useful outcome of the modelling exercise, variations in state-variables over space and time need to be considered and realistic representations of internal flow processes have to be computed.

Different types of models, categorized as physically-based and fully distributed, have been developed and successfully applied to describe individual processes of the hydrological cycle. Two important examples of such models are: soil water flow models, e.g. based on the one-dimensional Richard's equation, to simulate soil moisture conditions in a profile, and groundwater models for simulating groundwater flow and head in aquifer systems. When it comes to provide an integrated description over catchment areas, there seems to be strong and diverse opinions among theoretical oriented people about the validity and appropriateness of such physically-based and fully distributed model codes. A general argument put forward is that the spatial resolution used to represent flow processes in the models are only valid at small scale and the variations which can be accounted for in the models at catchment scale is far too coarse compared to the natural conditions. The results are therefore flawed to an extend that the reliability of the simulation results may be questionable.

Despite theoretical and philosophical discussions of what type of codes should be defined as physically-based model codes, and when and how they can be used (e.g. Refsgaard et al., 1995b), professionals involved with water resources problems recognize that the hydrological issues they are concerned with are introduced in a spatially distributed manner, and a conceptualization which mirrors the prototype conditions as closely as possible is required.

This is supported by the fact that groundwater model codes, which generally are accepted as being 'physically-based' models have been successfully applied in thousands

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of projects around the world in the past twenty five years. Even though many of these model applications have included crude approximate descriptions of the hydrogeological settings and the groundwater flow, for example three-dimensional flow regimes have been treated as two-dimensional, they have served the purpose to provide valuable information for decision making in connection with groundwater planning, management and protection.

One particular problem, which many of the users of these 'traditional' groundwater codes have experienced, is to define realistic boundary conditions, for example temporal and spatial pattern of groundwater recharge, flow exchange with rivers and channels systems, and appropriate dynamic conditions in areas with shallow water table. It is under such circumstances that integration of processes, covering surface water as well as subsurface water becomes important.

This Chapter gives a brief presentation of the concept and use of physically-based distributed models. We will use MIKE SHE as an example, but many of the conclusions drawn may be equally representative for other types of distributed model codes. A brief presentation of the hydrological processes in MIKE SHE will be given, and we will share some of our experiences in using MIKE SHE in connection with different types of water resources projects.

2. The MIKE SHE hydrological modelling system

Although it is nearly twenty years since the development of the Système Hydrologique Européen - SHE (Abbott et al., 1986a,b) was initiated, the model code MIKE SHE (a further development of SHE, (Refsgaard and Storm, 1995) is today one of the few catchment modelling codes available for project work, which may be categorized as a physically-based and fully distributed hydrological model code. A large number of other model codes have been developed, but are still mainly applied in research context, e.g. IHDM (Beven, 1985; Calver, 1988; Beven and Binley, 1992) and SWAGSIM (Prathapar et al., 1995). Despite discussions of the limitations and validity of e.g. MIKE SHE (Grayson et al., 1992; Smith et al., 1994), this modelling system has obtained a successful record of applications within the last few years.

MIKE SHE was developed as an alternative to the lumped conceptual rainfall-runoff models (e.g. Stanford model (Crawford and Linsley, 1966), NAM model (Nielsen and Hansen, 1973) etc.) to provide a rigorous approach based on accepted theories of the physically processes of surface and subsurface water and solutes. However, its current use in water resources and environmental projects has in many cases been generated from a wish to overcome some of the above mentioned difficulties that arise from using traditional groundwater models. In fact there is a great tendency among model developers to improve 'traditional' groundwater model codes to accommodate features that are included in the MIKE SHE.

The experience of the authors from using MIKE SHE in local and regional catchment studies under various data availability has been that the physically-based description provides an excellent framework for investigating a range of water resources problems

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on different temporal and spatial scales. Most of the process descriptions, which theoretically only are fully valid at small scale, provides often an acceptable conceptual framework on regional scale.

It is important to notice that although all natural systems exhibit pronounced spatial heterogeneity to an extent we never can hope to describe (neither in the model nor in the measured data), the underlying processes are often satisfactorily valid. However, it is the responsibility of modellers to ensure that approximations introduced in modelling applications, in terms of conceptualization (ie. description of the natural system), calibration accuracy, and predictions, are adequately reported and give advise on the limitations and the confidence we should put into the simulation results.

3. Description of Water Flow Processes

The basic description of the water flow in the surface- and subsurface water systems has changed very little from the original SHE code (Abbott et al., 1986a,b). The international collaboration effort in connection with its development made it necessary to design a modular programme structure comprising six individual process-oriented components, each describing one major hydrological process in the hydrological cycle. Individually, they can describe parts of the hydrological cycle, but combined, they provide a complete integrated description of the land phase part of the hydrological cycle, Fig. 1.

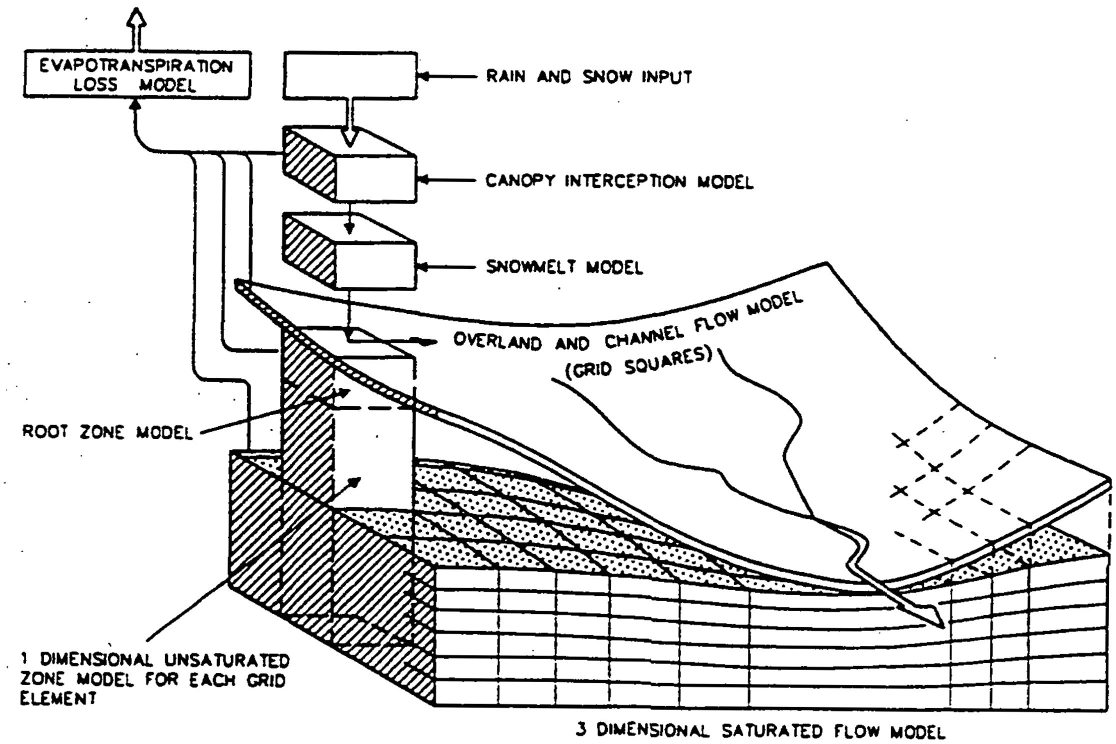

Figure 1 Schematic representation of the flow components of MIKE SHE

This modular structure has today become an important feature because many studies only require use of some of the process components. This may be relevant in studies where either data availability, time-frame or study objective does not justify a complete integrated description.

During the initial testing, and subsequent use in project work some of the process descriptions have been modified to enhance numerical efficiency, and accomodate new flow features, e.g. fractured flow (Brettman et al., 1993). However, an important part of the software development was devoted to produce an extensive user-interfaces to facilitate data processing and presentation of simulation results, but also to make the software more transferable, which some see as a potential risk for misuse (Grayson et al., 1992).

MIKE SHE was originally developed with a view to describe the entire land phase of the hydrological cycle in a given catchment with a level of detail sufficiently fine to be able to claim a physically-based concept. The equations used in the model are with few exceptions non-empirical and well-known to represent the physical processes at the appropriate scales in the different parts of the hydrological cycle. The parameters in these equations can be obtained from measurements as long as they are compatible with the representative volumes for which the equations are derived and used on the appropriate scale in the model application.

The flow processes represented in MIKE SHE and other physically-based distributed modelling systems include: snow melt, interception and evapotranspiration, overland and channel flow, vertical flow in the unsaturated zone, and groundwater flow. In MIKE SHE, individual processes operates at time steps consistent with their own temporal scales. For example, unsaturated flow cannot be expected to realistically represent the infiltration process and movement of sharp wetting fronts in the root zone if calculated at time steps of days or weeks which may be appropriate for groundwater flow. However, because of the computational demands of distributed models it is important to use as large time steps as possible, and this is often dictated by the actual hydrological conditions.

Below is given a brief description of the individual processes. For a more detailed review see Refsgaard and Storm (1995).

Variations in hydraulic heads, flows and water storages on the ground surface, in rivers and in the unsaturated and saturated zones are modelled in a network of grid squares. All spatial variations in input data and catchment characteristics can be represented by the spatial resolution given by the modeller.

Flow in the unsaturated zone is commonly described by the one-dimensional Richard's equation and solved in a finite difference scheme in the vertical as illustrated in Fig. 2, (Jensen, 1983). A similar approach is used in other unsaturated zone model codes using Richard's equation, e.g. SWATRE (Feddes, 1988). In MIKE SHE the solution technique has an advanced feature which estimates the exchange of water between the saturated zone and the unsaturated zone based on the retention, and adjusts the phreatic surface accordingly (Storm, 1991). The simulation of the unsaturated flow plays a central part in most model applications, e.g. recharge estimation, transpiration, surface-groundwater interaction and fate of pollutants.

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The use of Richard's equation is probably the most controversial issue in connection with catchment application, and in many investigations this approach would not be necessary. SWAGSIM (Prathapar et al., 1995) uses an analytical solution to calculate the flow in the unsaturated zone, which provides a much faster model use. Today alternative versions, e.g. neglecting the capillary forces, are included in MIKE SHE. which under many conditions provide sufficient accurate estimates and considerable computational savings.

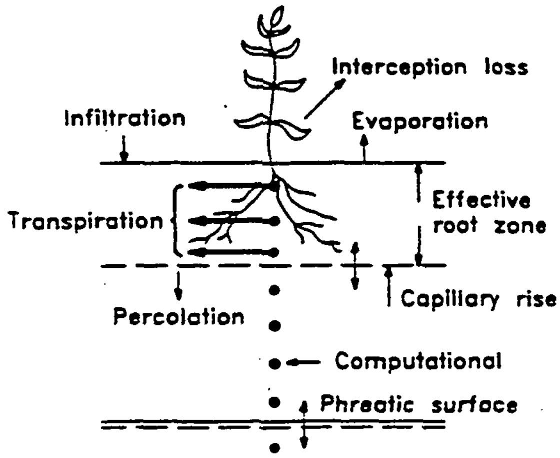

Figure 2 Illustration of the node representation for the unsaturated zone

Actual evapotranspiration is calculated from potential evaporation data. Two methods are included, which are either based on an empirical formula (Kristensen and Jensen, 1975) or on the Penman-Monteith Equation (Monteith, 1965). Both methods use the actual soil moisture/retention conditions in the root zone to calculate the actual evapotranspiration loss. The amount of water that can be drawn out of the root zone depend on crop and soil properties. The interception/evapotranspiration component is an integral part of the unsaturated zone component to determine the timing and magnitude of recharge and overland flow generation.

Overland flow is generated when the top-soil becomes saturated. The routing of the water is computed using a two-dimensional diffusive wave approximation of St. Venant's equation (Preissman and Zaoui, 1979). Net rainfall, evaporation and infiltration are introduced as source/sinks allowing the surface to dry out on more permeable soil areas. The solution assumes sheet flow conditions, which under regional applications is a very crude approximation recognised for example when MIKE SHE provide the framework for soil erosion modelling (Styczen and Nielsen, 1989; Hasholt and Styczen, 1993). Local depressions in the ground surface and physical barriers (e.g. levees, roads etc.) are conceptually modelled using a detention storage allowing water only to evaporate or infiltrate as long as the water level is below a specified threshold.

Routed excess water on the ground surface enters the river system as lateral inflow. Flow in channels and streams is simulated using a branched and looped one-dimensional diffusive wave approximation of the St Venant equation. The representation of the river in the models is approximated to run along the boundaries of individual grid squares.

Groundwater flow is calculated from a three-dimensional governing equation (Refsgaard and Storm, 1995). The equation is solved numerically by a finite difference method using a modified Gauss-Seidel implicit, iterative scheme (Thomas, 1973). The spatial resolution in the vertical can either follow a rigid network for fully three-dimensional flow or follow the geological layering for a quasi-three-dimensional (or in special cases two-dimensional) flow. Discharge to or recharge from the river system occurs from all computational cells located along the river links. The integrated description provide a more comprehensive interaction with the river, accounting for the dynamic variations in river water level. A feature which is often lacking in traditional groundwater models, e.g. MODFLOW (McDonald and Harbaugh, 1988), etc.

3. Description of solute transport processes

Since the initial development of SHE more attention has been given to water quality aspects, especially in the groundwater used for water supply purposes. Leakage of pollution from waste disposal sites, polluting industries, and agricultural areas etc. has increased the need for modelling tools which can assist in detection and cleaning up of contaminated areas. Transport models are important tools for prediction of future likely pollution patterns, designing optimal remediation schemes, or designing water quality monitoring network. Models to locate new waste disposal sites to prevent further deterioration of the water quality in water supply areas is another important aspect.

Water quality issues are not confined to the groundwater only, but often experienced in soils and receiving rivers as well. The interaction between surface water and subsurface water becomes therefore an important aspect also in many environmental problems. For example infiltration of poor water quality river water may lead to problems in connection with river bank filtration schemes for water supply, or use of fertilizer can deteriorate the water quality of the groundwater and the receiving rivers dramatically.

The advection-dispersion module developed as an add-on module for MIKE SHE can describe the transport processes for all the water flow components in MIKE SHE. This means that transport of solvents can be simulated for the entire hydrological cycle in a catchment as illustrated in Fig. 3. This is important in the above mentioned cases, but provide also a powerful option in connection with e.g. groundwater management. A regional model of the hydrological system can be developed and form basis for detailed local models including solute transport simulation, (Refsgaard et al., 1994).

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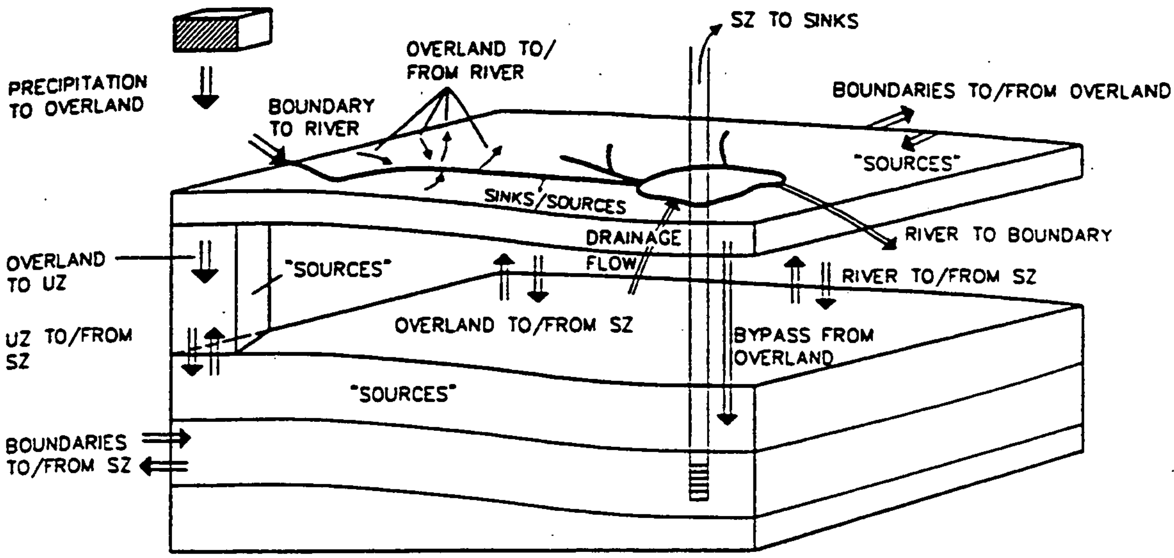

Figure 3 Schematic representation of the transport components of MIKE SHE

Basically or microscopically, solute transport in porous media is governed by three processes: advection, molecular diffusion and kinematic or mechanical dispersion. Advection describes the movement of solutes carried by the flowing water which is the flow velocities computed by the water flow module in MIKE SHE. As a consequence of this advection-dispersion simulations reflect to a large degree the uncertainties and errors in the flow description. Diffusion describes the spreading of solute molecules by virtue of their kinetic motion the diffusive flux normally being described as proportional to the concentration gradients. The kinematic dispersion is the spreading of the solutes caused by microscopic variations in the flow velocity field which usually is not represented by the spatial resolution used in the models.

The governing equations used in the different components of MIKE SHE are similar to the ones normally used in transport models. They are all based on the general advection-dispersion equation (ADE), see e.g. Bear and Verruijt (1987). The difference between the numerical equations is reflected in the dimensionality, the formulation of the dispersion coefficients and the numerical method applied.

Most attention has been given the groundwater transport component as it includes both a solution to the three-dimensional formulation of the ADE, a particle tracking solution, and an option for modelling transport in a dual porosity medium, (Brettmann et al., 1993). The dispersion term in the groundwater component includes options for isotropic conditions and anisotropic conditions with axial symmetry around the vertical axis, see Bear and Verruijt (1987). The ADE is solved using an explicit scheme called the QUICKEST scheme (Vested et al., 1992). The solute transport in the unsaturated zone is solved using a one-dimensional version.

Transport in water on the ground surface is computed using a two-dimensional formulation of the ADE. Solutes in rain and exchanges of water and solutes with the

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unsaturated zone and the groundwater (e.g. seepage) are introduced as sources/sinks. Evaporation of ponded water may lead to high concentrations and solutes are allowed to precipitate if the concentration exceeds a certain level and dissolve again when the dilution becomes high enough again. The ADE is also solved using the QUICKEST scheme.

In the channel transport component, the ADE is simplified to a one-dimensional formulation. Lateral inflow from the overland and base flow/leakage loss are treated as source/sinks. The equation is solved using a modified Lax scheme (Abbott and Cunge. 1982).

4. Types of application

A number of possible application areas where physically-based distributed models codes would be applicable and provide a better approach than traditional lumped or semi-distributed conceptual models have been suggested by e.g. Abbott et al. (1986a,b), Bathurst and O'Connell (1986, 1992). Many of those have been demonstrated in subsequent studies as described below. Others, for example prediction of impacts of land-use changes, runoff prediction from ungauged catchment, still need to be shown. In fact, the above references may have been too optimistic regarding the direct applicability of point measurements to estimate model parameter values and our ability to relate changes in catchment conditions to changes in the parameter values used in model codes.

Refsgaard et al. (1995b) showed that focusing on only runoff simulation from either gauged or ungauged catchments, a physically-based model (MIKE SHE) did not prove significantly better than a simple lumped conceptual model - NAM. For ungauged conditions, the physically-based model may narrow the error-band of the prediction, but considering the effort required to set up the model, the simple lumped conceptual model would normally be the most appropriate.

Bathurst and O'Connell (1992) found that for event modelling, physically-based approaches such as SHE provides a reliable basis for model prediction as soon as a short period (single event) is available for model calibration. However, our findings suggest that this may not be generally true. From comparisons of long time series of runoff, there seems no evidence that the rigorous approach provide significantly better prediction than a simple lumped conceptual model, unless there are features in the runoff simulation which is neglected by the lumped conceptual model. The latter was demonstrated on some stream systems in connection with simulation of summer runoff, where a substantial loss from the stream occurred due to potential evapotranspiration rates from low-lying areas close to the stream. The ability of MIKE SHE to account for the spatial variations in depths to groundwater table was important for calculating the capillary rise in these areas.

It is also our experience that prediction of variables which are not directly tested (calibrated against) may include considerable uncertainties. It should be stressed, that this does not limit the applicability of physically-based distributed models, and as

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illustrated below, there is a wide range of water resources investigations, for which this type of modelling is not only warranted but also the only alternative. This concerns in particular analysis of localised human activities such as groundwater abstraction schemes in environments where the interaction between surface water and subsurface water is a crucial issue.

A large number of applications confirm the wide range of water resources problems for which a model code like MIKE SHE is a suitable tool. They also reveal that most of the studies have focused on the groundwater system, and the benefits from choosing MIKE SHE was primarily the ability to integrate the groundwater flow with the surface water system. It is worth mentioning that a large number of other MIKE SHE applications, which are not directly known to the authors, have been carried out, mainly in engineering projects.

Early applications of MIKE SHE were mainly concerned with runoff predictions, see e.g. Jain et al. (1992) and Refsgaard et al. (1992). The model applications were part of a technology transfer project, and gave important operational experience in simulations on medium-scale catchments (up to 5000 km^2). In general, limited data were available, obtained mainly from literature, but as part of the project work, data became available through dedicated field investigations to enhance the model accuracy and reliability.

A number of studies have been concerned with groundwater development and its impact on groundwater heads and river environment (Refsgaard et al., 1994; Refsgaard and Sørensen, 1994; and Refsgaard, 1994). These models generally covers large multi-layered aquifer systems, where the interaction between the layers need to be considered. The projects have provided a framework for managing the groundwater and protect the surface water environment. In some of the studies, a proper allocation of pumping sites needed to be addressed. In broader environmental impact assessments, the regional models have in some cases also provided boundary conditions for local detailed models to examine e.g. the threat of contamination of water supply wells from waste deposits. Styczen and Storm (1993, 1995), presented a methodology for using MIKE SHE in connection with non-point agricultural pollution.

Recently, MIKE SHE has been introduced in irrigation and salinity planning and management projects. Mudgeway and Nathan (1993) used it to simulate flow and salt transport processes, over a three months period, between shallow groundwater and surface waters in a 8,9 ha set of irrigation bays in the Tragowel Plains in Victoria Australia. The model was tested against drain flow, groundwater table, soil moisture and salt concentrations. The simulations were used to study the effect of deepening drains on discharge of salt loads.

Storm et al. (1996) repeated the simulations using a much longer time series (eighteen months) of observations. The model was calibrated against drain flow (two stations), groundwater levels (five piezometers) and soil moisture at eight sampling points. The simulation exercise demonstrated an interesting point, namely that calibration against selected state-variables such as drain flow and/or groundwater levels alone, may not necessarily lead to a correct predictions of other state-variables, e.g. soil moisture conditions even though the data availability for parameter assessment may be

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quite comprehensive.

In a similar study, but on regional scale (approx. 2700 km^2), Storm and Punthakey (1995) used MIKE SHE to evaluate a number of options proposed in a Land and Water Management Plan for the Wakool Irrigation District (WID) in NSW, Australia. The WID experiences, as many other large irrigation districts in semi-arid regions in Australia, an increase in the area with shallow saline groundwater. MIKE SHE provided a complete description of the complex hydrological regime in WID involving temporal and spatial variations in the exchange of water between the ground surface, drainage and supply systems, and the groundwater aquifers within the area. Management options to be analyzed for a time frame of 30 years and included scenarios which focused on the surface water regime (extension of the drainage system and/or sealing of supply channels where seepage losses were observed) as well as the subsurface water regime (groundwater pumping schemes in shallow and deep aquifers).

Figs. 4 a-c show the estimated progress of shallow water table areas for two scenarios, a so-called 'no plan' option, where no additional action is taking place and one where additional 48 pumps are installed in the shallow aquifer system.

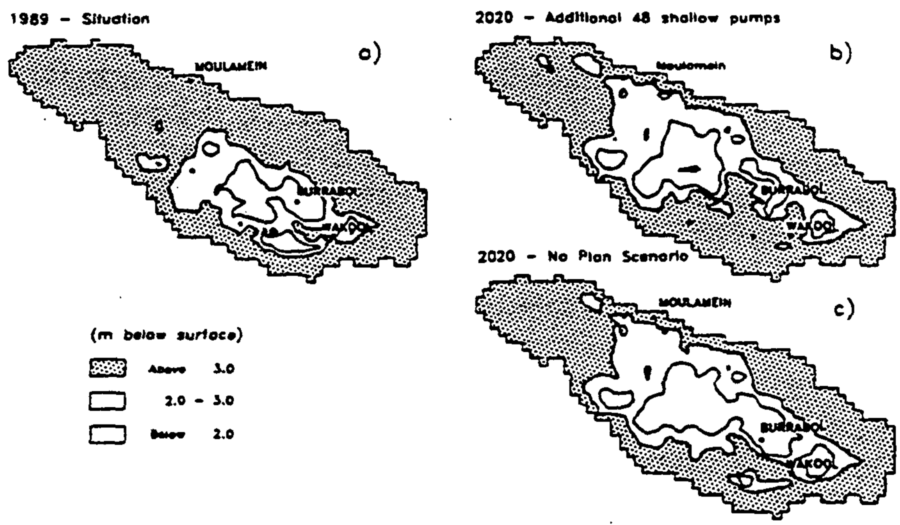

Figure 4 Areas of shallow water table in 1989 and predicted for year for two scenarios

Refsgaard et al. (1995a) used MIKE SHE to evaluate the efficiency of a pump-and-treat system for remediating a severe Chlorinated Carbon Solvent contamination. Model results showed that the pump-and-treat system would not lead to significantly improvements of conditions in the groundwater aquifer and certainly not prevent contamination of the recipient located about 2 kilometres down gradient from the pollution source. The conclusions lead to a close down of the remediation pumping and establishment of a monitoring system to ensure reasonable conditions in the recipient in the future.

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5. Problem areas using physically-based distributed models

In most catchment studies carried out so far, it was not possible to use spatial resolution sufficiently detailed to claim that the developed models were physically-based (not the same as claiming the model code to be physically-based). In fact, it was realised from early test applications that considerable lumping is involved at the scale of a grid square. This is not a new experience, but have been recognised by modellers working with groundwater modelling. They have successfully been using transmissivities and storage coefficients derived from e.g. pumping tests. It is evident to everyone that such parameter values are gross simplifications of actual aquifer conditions, but even then many professionals have found great benefits from using groundwater models for different types of problems.

It is the authors experience that the spatial resolutions and variations in properties used, can provide a good representation of the conditions of the area, and account for significant differences in e.g. recharge pattern across the model areas. In practise, spatial variations in catchment characteristics are often obtained from maps describing topography, soil and land-use pattern and interpreted geological conditions, combined with information about representative properties for different map units.

Parameters values obtained from the above information are only initial estimates, which are modified during the model calibration, where simulated state-variables are matching observed conditions at discrete points. The final parameter set for a specific model, is subjective and the outcome of several factors including the modellers perception of the catchment and flow processes, the spatial resolution used, the degree of matching of observed and simulated variables obtained etc.

It is important to emphasize that there is a number of fundamental scale problems which needs to be carefully considered in all model applications involving distributed models. This becomes particular important when considering interactions between the surface flow and the subsurface flows. A few areas where we have encountered scale problems include:

* The flow exchange between groundwater aquifer and river system. Since the flow is based on Darcy's law using the gradient between the river water level and the groundwater heads in the adjacent grid squares, the flow rates and the resultant head changes will depend on the spatial resolution used. This is an important aspect in for example simulating the hydrograph recessions correct.

* In catchments with a dense drainage network it is often not possible to represent entire drainage system (many streams are of ephemeral nature). For such situations sub grid variations in the topography need to be accounted for in order to simulate the hydrograph response in the main streams correctly.

* Simulation of infiltration and vertical unsaturated flow in the soil. the hydraulic parameters used in Richard's equation can be obtained from laboratory measurements at small undisturbed soil samples. However, for grid squares covering large areas

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(e.g. 25 hectares) these are seldom representative unless completely homogeneous conditions exist in the horizontal directions. Therefore effective or representative parameters are used, which means that the simulated soil moisture conditions can not be validated directly.

The latter point is often subject to more discussion of MIKE SHE than all the other issues. In most catchment simulations the use of Richard's equation becomes conceptual rather than physically-based and simpler approaches could be chosen. Nevertheless, this equation provides a good routing description, and the capability to simulate capillary rise under shallow water table conditions is important in studies dealing with wetland areas or irrigation in shallow water table area. For situations where we know that Darcy's law is not applicable, empirical formulas describing flow in preferential flow paths is included in the simulation. The similar problem concerning infiltration in cracking soils was demonstrated by Storm et al. (1996), where an empirical cracking option was necessary to describe high infiltration rates in clayey soils during early parts of irrigation events.

Because representative parameter values are used, the certainty in the model results depends on the data available to compare the simulated spatial and temporal variations with observations. Again, this is well-known from groundwater model applications, where the model parameters (conductivities or transmissivities) are estimated from the calibration against observed head variations in discrete points.

For regional catchment studies the model performance is usually evaluated based on comparisons against river discharges and groundwater heads. Very seldom measured soil moisture data are available, and if they are such comparisons will require that the site specific properties are known.

It is often stated that distributed models requires a large number of data and therefore very time consuming and complicated to set up and calibrate. In fact, a number of short-term screening evaluation projects have been carried out with MIKE SHE for example in connection with studying the contamination risks from waste disposals. In these studies only sparse existing information about the hydrogeological conditions was available. The model was used to obtain an improved knowledge about the possible flow patterns around the waste disposal site based on the existing geological interpretations. These applications could also be used to identify where existing knowledge is lacking and assist in defining appropriate monitoring programmes.

Another common argument against fully distributed models is the risk of over-parameterization. This risk is of course always there. However, the general experience is that if the data are lacking to describe the spatial variations in the catchment it is too time-consuming and not worthwhile to modify a large number of parameter values in order to improve e.g. hydrograph predictions. In such cases very few parameters are modified during the calibration and the reliability of the results are evaluated with this in mind.

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6. Conclusions

The recent experience with the use of MIKE SHE in a range of water resources projects show that physically-based and fully distributed model codes such as MIKE SHE are important tools for assessing different types of water planning and management issues. Although it can be argued that the governing equations used in this type of modelling systems not always are fully valid on the scale they are applied, they provide an important flexibility to investigate problems on different scales.

MIKE SHE has been used in small-scale research studies (where the equations are directly valid) as well as large scale catchment projects, where considerable lumping and conceptualisation is imposed. In the latter type of studies, the models are calibrated in accordance with the common practice as made in for example groundwater studies. It is important in this type of application to recognise the general limitations of the developed model depending of the various assumptions used in the conceptualisation, calibration and prediction phases.

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CHAPTER 6
SOIL EROSION MODELLING

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1. Introduction

Increasing rates of soil erosion in developing countries have been given attention for a long time. The increase in population pressure, inequality in societies, and sometimes also legislation have resulted in cultivation of areas unsuitable for crop production or in unsustainable farming which, together with overgrazing, are major reasons for soil erosion. Also erratic rainfall results in ecosystems prone to erosion, in particular in the semi-arid regions where the amount of rainfall impedes the establishment of good ground cover. Whitlow (1988) has estimated that average soil losses on croplands and grazing areas on Communal Lands in Zimbabwe are 50 and 75 t/ha/year, respectively, whereas the rates of soil formation are very slow, e.g. 400 kg/ha/year.

Moreover, soil erosion is increasingly being recognized as a hazard in European countries, in particular in the Mediterranean area and on the loamy, sandy loamy, and sandy soils of northern Europe. In Belgium and England measured erosion rates from bare ground were in the range 7-82 t/ha/year (Bollinne, 1978) and 10-45 t/ha/year (Morgan, 1985), which is far above the soil loss tolerance of 1 t/ha/year for northern Europe as suggested by Evans (1981).

Oldeman (1992) estimated that worldwide 24 percent of the inhabited land area are affected by man-induced soil degradation ranging from 12 percent i North America to 27 percent in Africa and 31 percent in Asia.

The harms of erosion are twofold. At the location where erosion takes place, infiltration rates, crop production and often the waterholding capacity as well are reduced through the removal of organic matter and plant nutrients. Furthermore, the transported material causes decreasing water quality, increasing eutrophication, and reduced life time of reservoirs due to siltation.

Much work has been put into development of soil erosion models over the last years to obtain a good tool for evaluation of soil erosion problems. Models are expected to assist in the following fields:

(a) Assessment of the extent of soil and nutrient losses and sediment transport in various environments.
(b) Land use planning as they can provide important information on the effects of changes in land use and of implementation of different soil conservation measures on soil losses and sediment yields.

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(c) A better understanding of the erosion processes, the dynamic and relative importance of the single processes and their interactions. Thus, just as model development relies on the research on erosion processes, models are important tools to test new findings in soil erosion research.

While the first soil erosion models were empirically-based, much of the recent work is now concentrated on the development of more physically-based descriptions of processes and their interactions. Simultaneously, there is a trend towards greater interaction between researchers involved in experimental work, theory development and modelling. This is a good development, because model building is probably the strongest tool available for evaluation of the relative significance of different processes, for evaluation of the sensitivity of the system to different interventions, and for discovering new angles to the given problem.

The present chapter will be restricted to erosion caused by water before it reaches a river or stream. Furthermore, the text will be confined to modelling of rill and interrill (sheet) erosion, and therefore does not include soil loss due to gully erosion and landslides.

Section 2 gives a short discussion of the various types of soil erosion models: from the earlier empirically-based, and mathematically simpler versions to the distributed physically-based models based on the recent research. The data input requirements and advantages and limitations of the various types of models in relation to different types of applications are briefly discussed.

The rest of the chapter focuses on physically-based distributed soil erosion modelling. The model development is discussed in Section 3 emphasizing the soil erosion processes and variables to be included. The requirements to the associated hydrological models as well as the linkages between the erosion and hydrological models are briefly discussed, too.

The construction, calibration, and validation of soil erosion models are shortly discussed in Section 4, while Section 5 contains a case study on the application of EUROSEM/MIKE SHE, a distributed physically-based soil erosion model.

Finally, constraints in soil erosion modelling, the possible application of physically-based models on a catchment scale and future research needs are summarized in Section 6.

2. Classification of Soil Erosion Models

A number of models, of various complexity, has been developed in the past. Like hydrological models soil erosion modelling has moved from empirically-based and simple mathematical models, e.g. the Universal Soil Loss Equation, USLE (Wischmeier and Smith, 1965) towards physically-based and mathematically much more complicated models like the European Soil Erosion Model, EUROSEM (Morgan et al., 1995).

Basically, there are three categories of soil erosion models: 1) Empirical, 2) Conceptual (or partly empirical/mixed), and 3) Physically-based. In Table 1 some of the most used models within each of the three categories are listed.

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TABLE 1. List and key characteristics of a number of the most used soil erosion models.

Model name(s)	Type of model	Scale of application	Temporal resolution	Spatial resolution	Separate rill/inter-rill components	Event-based/ continuous	References
USLE and RUSLE	Empirical	Hillslope	Yearly soil loss	No	No	-	Wischmeier & Smith, 1978; Renard et al., 1994
SLEMSA	Empirical	Between ridges	Yearly soil loss	No	No	-	Elwell, 1978
ANSWERS	Conceptual	Catchment	Distributed	Distributed(2-D)	No	Event-based	Beasley et al., 1980
CREAMS	Conceptual	Field-scale	Total storm loss	No	Yes	Event-based	USDA, 1980
Calvin Rose	Physically-based	Plane element, e.g. uniform slopes	Distributed	Distributed(1-D)	No	Event-based	Rose et al., 1983a,b
SEM	Physically-based	Catchment	Distributed	Distributed(2-D)	Yes (for hillslope)	Event-based (continuous)	Nielsen and Styczen, 1986 DHI and IoG, 1992
WEPP	Physically-based	Hillslope version Catchment version	Distributed	Distributed(1-D) Distributed(2-D)	Yes	Continuous	Lane and Nearing, 1989
EUROSEM/ KINEROS	Physically-based	Individual fields and small sub-catchments	Distributed	Distributed(2-D)	Yes	Event-based	Morgan et al., 1995
EUROSEM/ MIKE SHE	Physically-based	Hillslopes and small catchments	Distributed	Distributed(2-D)	Yes	Continuous	DHI (1994)
SHESED-UK	Physically-based	Small sub-catchments	Distributed	Distributed(2-D)	No	Continuous	Wicks et al. (1992)

2.1. EMPIRICAL MODELS

Most of the *empirical models* are based on data from field observations, mostly standard runoff plots on uniform slopes, and are usually statistical in nature. The first empirical model to be developed was the Universal Soil Loss Equation, USLE, (Wischmeier and Smith, 1965, 1978), which also is the most well-known and most widely used empirical model. Although highly criticized - and for good reasons - it is still in use and has undergone a number of modifications. Thus a Revised USLE, RUSLE, to be run on a computer has been developed recently. The USLE predicts the annual soil loss from small areas on a slope, and the RUSLE maintains the basic structure of the USLE, namely

$$A = R\,K\,L\,S\,C\,P \tag{1}$$

where A is the computed annual soil loss, R is the rainfall-runoff erosivity factor, K is the soil erodibility factor, L is the slope length factor, S is the slope steepness factor, C is the cover-management factor, and P is the supporting practices factor (Renard et al., 1994).

A similar model, the Soil Loss Estimation Model for Southern Africa, SLEMSA, primarily based of field plot erosion studies in Zimbabwe, has been developed by Elwell (1977).

The main limitation of USLE/RUSLE and empirical models in general, is their limited applicability outside the range of conditions for which they have been developed. Adaption of e.g. the USLE to a new environment requires a major investment of resources and time to develop the database required to drive the model (Nearing et al., 1994).

Although the empirical models may give reasonable estimates of annual soil loss from a field, they are not adequate for catchment scale estimations. For example, they do not take into account deposition at the lower parts of a hillslope, which is relevant in relation to transport of sediment and pollutants to rivers and reservoirs. The models estimate the annual soil loss and can not therefore be used to study the temporal dynamics of erosion. The empirical models only provide a limited insight in the relative importance of the various variables and their sensitivity in different environments. USLE suffers from the conceptual defect that rainfall and soil factors (among others) cannot simply be multiplied because of the subtractive effect of soil infiltration capacity in generating erosive runoff from a given rainfall (Kirkby, 1980).

A modified version of the USLE, the MUSLE (Williams, 1975) has tried to overcome a number of the above-mentioned problems by introducing an empirical runoff energy factor instead of the rainfall factor, and the model is able to estimate sediment yield from single storms.

2.2. CONCEPTUAL MODELS

Realisation of the insufficiency in the application of the USLE led to the development of a number of *conceptual models* in the 1970s, including CREAMS (USDA, 1980),

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ANSWERS (Beasley et al., 1980) and the modified ANSWERS, MODANSW (Park et al., 1982).

These models lie somewhere between the empirically and physically-based models. The main step forward by the development of these models was the introduction of the laws of conservation of mass and energy, i.e. the continuity equation and the grouping of the area of concern into a number of elements/grids in order to describe the spatial variations in erosion and deposition. The detachment and transport of sediment from each element/grid follow the model proposed by Meyer and Wischmeier (1969). The outflux of sediment from a given grid/element is determined by the influx of sediment plus the net detachment of sediment by runoff and rainfall within the element/grid with the maximum limit that the outflux never exceeds the total transport capacity. Thus, regarding these basic concepts these models resemble the physically-based distributed models.

On the other hand, for a number of processes described in the models USLE factors are used, and their physical validity is in some cases questionable. Both the C and K factors developed for USLE refer to total soil loss, and one cannot expect these factors to represent the individual processes in a single storm as is done in ANSWERS and MODANSW. Likewise, some of the processes in CREAMS are questionable. For example, the model calculates detachment on a storm basis whereas sediment transport is calculated on an instantaneous rate basis.

Thus, the main limitations of the conceptual models lie in the poor physical description of the processes which, among other things, results in distortion of parameter values determined by calibration (Elliot et al., 1994).

2.3. PHYSICALLY-BASED MODELS

During the last 10-15 years most of the work on soil erosion modelling has been concentrated on the development of *physically-based erosion models*. The physically-based models are intended to represent the essential mechanisms controlling erosion. The models include most of the factors affecting erosion and their spatial and temporal variability, and the subprocesses and their complex interactions are described as well.

Three physically-based models will be mentioned here: 1) WEPP (USDA Water Erosion Prediction Project) which mainly is based on research results from the USA (Lane and Nearing, 1989; Nearing et al., 1989), 2) EUROSEM (European Soil Erosion Model) which mainly is based on recent soil erosion research in Europe (Morgan, et al., 1995), and 3) the soil erosion model developed by a group of Australian scientists (Rose et al., 1983a,b).

The basic erosion concepts of most of the physically-based models are rather similar, whereas the way they are linked to a hydrological model and the use of equations to model the individual processes vary. All of the above-mentioned models have separate interrill and rill components. Rill erosion is described as a function of the flow's ability to detach sediment, of the sediment transport capacity, and of the existing sediment load in the flow. The models use different transport capacity equations as well as different thresholds for rill initiation. Also, the detachment and transport in interrill areas are modelled in a different way.

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It should be stressed that the most important basis for the physically-based soil erosion models is an adequately distributed simulation of the driving variable in the soil erosion and transport processes - namely the overland flow. This is in particular important when it comes to a more precise description of rill initiation and development. Thus, the linkage of the erosion model to a distributed hydrological model with a comprehensive overland flow component is a prerequisite.

The EUROSEM model is the most recent as well as the most comprehensive physically-based distributed model. It includes effects of plant cover on interception and rainfall energy; rock fragment (stoniness) effects on infiltration, flow velocity and splash erosion, and the changes in the shape and size of rill channels as a result of erosion and deposition (Morgan et al., 1995).

The physically-based distributed soil erosion models have a number of advantages as compared to the empirical and conceptual models. Due to a physically-based description of the processes research scientists can use the models to test new theories, and sensitivity analysis can help identifying which factors or erosion processes are the most important to the overall erosion process and therefore should be given more attention in research. Due to the calculation of the spatial as well as the temporal variations of sediment concentration this type of models provides e.g. a much better possibility for identifying areas with high erosion risk and to extrapolate soil erosion rates from plot to catchment scale. Thus, such models are *potentially* much more useful as planning tools.

However, presently the large requirement to input data and computer power as well as the complexity of the models, restrict a wider application of the models.

3. Soil Erosion Processes in Physically-Based Models

3.1. MODELLING THE VARIOUS EROSION PROCESSES

A comprehensive outline of the state-of-knowledge of soil erosion processes used in modelling is outside the scope of this chapter. However, as physically-based soil erosion modelling generally is still at an early stage of development as compared to physically-based hydrological modelling and many of the processes are not yet well understood, the most crucial soil erosion processes, and in particular processes which need more attention, are discussed below.

Key soil erosion processes and their interactions as included in the European Soil Erosion Model, EUROSEM (Morgan et al., 1995) are shown in Fig. 1.

3.1.1. Soil Detachment By Raindrop Impact (Splash Erosion)

The two major variables in modelling the detachment by raindrop impact are the ability of the raindrop to detach the soil (rainfall erosivity) and the ability of the soil to resist the raindrop impact (soil erodibility). At least two other variables have to be considered, namely the vegetation canopy, which influences the raindrop diameter and velocity and the flow depth that may reduce/(dissipate) the erosivity of the rainfall.

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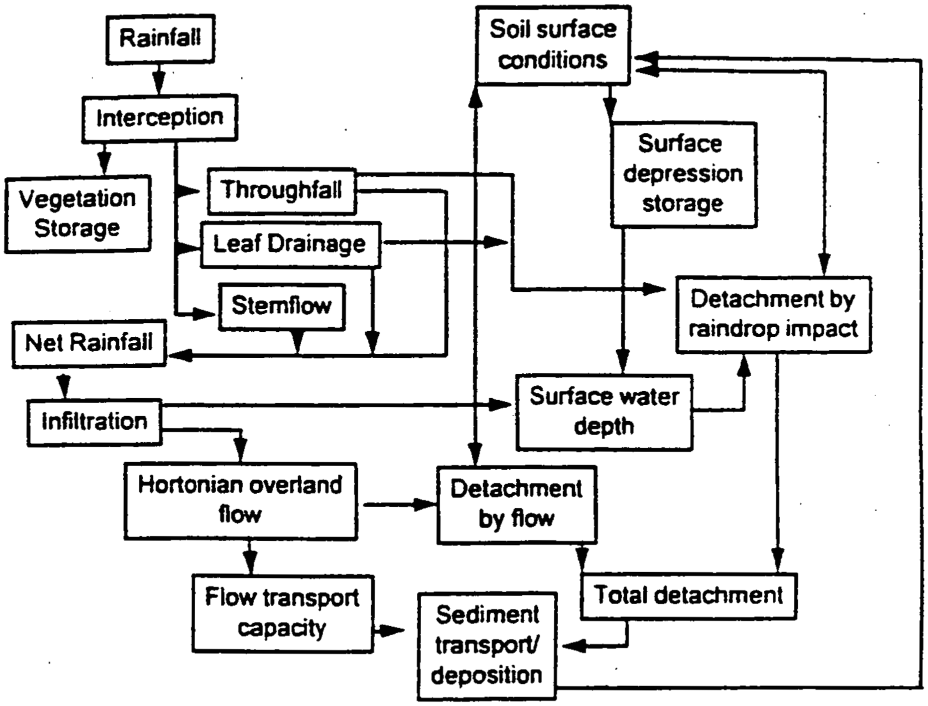

Figure 1. Flow chart of the European Soil Erosion Model, EUROSEM (Morgan et al., 1995).

It is generally accepted that rainfall is the main detaching agent in interrill areas (Foster et al.; 1982, Gilley et al., 1985), mainly because most rains strike the surface at velocities between 5 to 9 m/s, while runoff velocities usually are less than 1 m/s in sheet flow (Meyer, 1981). Most splash erosion models have been correlated to the energy, intensity or momentum of the rain. Morgan et al. (1995) prefers the expression:

$$DET = k\,KE\,e^{bh} \tag{2}$$

where DET is the soil detachment by raindrop impact (g m^{-2}), k is an index of the detachability of the soil (g J^{-1}), KE is the total kinetic energy of the rainfall at the ground surface (J m^{-2}), h is the depth of the surface water layer and b is an exponent varying between 0.9 and 3.1 (Torri et al., 1987b). As the soil parameters are poorly known, they are often used as a calibration factor.

If the soil is covered by vegetation, the estimation of splash erosion becomes more complicated. Different correction factors have been used, among these the C-factor from the USLE (Beasley et al., 1980), or the C_I, C_{II} and C_{III} factors, defined by Wischmeier (1975). However, these correction factors are always smaller than or equal to 1 and therefore fail to describe situations found by e.g. Mosley (1982), Morgan (1982) and Morgan et al. (1985) where more splash is measured under vegetation than on bare soil.

The theoretically based equation for splash erosion developed by Styczen and Høgh-Schmidt (1988) for cohesive soils can include effects of a canopy, as it considers splash

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erosion to be proportional to the sum of the squared momentum of each drop hitting the ground:

$$\text{Splash} = \frac{A\,Pr}{2\,\hat{e}} \sum_{i=0}^{D} N_D\, P_D \tag{3}$$

where Splash is the amount of splash erosion (kg m^{-2} s^{-1}), ê is the energy needed to break the bonds between two microaggregates (J), Pr is the probability of energy excess (energy left for lifting the particles), A is a soil parameter which is a function of the above probability function, N_D is the number of drops (m^{-2} s^{-1}) with the diameter D, and P_D is the squared momentum of a drop with the diameter D (m). Basically the factors before the summation sign are soil parameters and after the summation sign rainfall parameter (compare with eq. 2). In case of vegetation, the summation will include separate expressions for the throughfall and the leaf drip. For non-cohesive soils (ê = 0), the rainfall parameter becomes a kinetic energy expression instead of a squared momentum. Studies by Styczen and Høgh-Schmidt (1988) indicated that the use of squared momentum is in better agreement with experimental results with and without vegetation than when models based on energy or intensity are used.

The soil erodibility term influencing splash erosion has to be specifically related to forces that bind the soil mass together as in eq. 3. The soil shear strength, τ_s, is probably the best physical measure to represent these forces - e.g. Al-Durrah and Bradford (1981, 1982a,b) and Torri et al. (1987b) found good correlation between shear strength and amount of detached soil. Good correlation has also been found between splash erosion and aggregate stability (Bryan, 1976). Due to the lack of data the soil shear strength has also been related to the soil texture (e.g. Morgan et al., 1991). However, this does not allow for treating the erodibility as a temporal variable, and it does not include the importance of the organic matter content. On the other hand, much more research is needed before the variability of soil shear strength can be properly modelled (see also section 3.1.4).

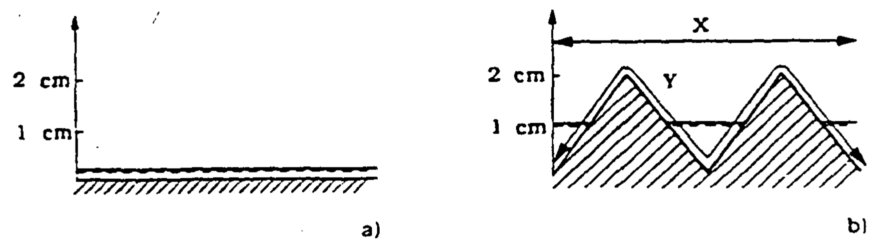

Figure 2a,b. Two soil surfaces having the same average flow depth but the effects of the water depth on the splash erosion are likely to be different.

The splash erosion decreases when the runoff depth increases. Most studies have shown that the splash erosion decreases exponentially with increasing depth, as shown in eq. 2.

However, eq. 2 applies to smooth surfaces, whereas on most surfaces in the field the water seldom is equally distributed. This is illustrated in Fig. 2a,b where two soils are

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having the same average water depth but are likely to experience different rates of detachment. Also here, a better description of the microtopography would help to estimate the effect of flow depths on splash erosion

3.1.2. Infiltration Conditions and Generation of Overland Flow

As overland flow is the major transporting agent and in some cases also the main detaching agent, a proper description in space and time of the generation and routing of overland flow is crucial.

Regarding the generation of overland flow the *infiltration rate* is the most sensitive variable. This is in particular true in cases where the rainfall intensity and the infiltration rate are of the same order of magnitude as this may give rise to simulation rates which -relatively - deviate considerably. To simulate the infiltration properly the model must include a sub-model for the unsaturated zone, and detailed input data on saturated hydraulic conductivity and soil moisture retention curves are therefore needed. The infiltration rate may be influenced by several other factors, such as frost/thaw, presence of stones and crusting.

Particularly in tropical environments, Hortonian type of overland flow may be an important generator of overland flow. In other environments, and in particular in the temperate regions, the major source of overland flow is saturated overland flow. In humid vegetated areas soil moisture levels tend to build up downslope, especially close to streams, and near-saturated areas generate a disproportionate amount of overland flow runoff (Kirkby, 1980). Thus, information on the spatial and temporal variation of depth to groundwater tables (primary or secondary) and description of existing semi-impermeable or impermeable layers are valuable in order to simulate the generation of saturated overland flow.

In the colder temperate regions erosion, and in particular rill erosion, is often caused by rainfall or snow melt on partly frozen soils. This is a process that is very difficult to model in details, e.g. because the depth of the snow can vary considerably due to local differences in wind and shelter conditions. Apart from snow melt, the issues to be modelled are the depth of thawed surface soil and estimation of shear strength which changes through the frost-thaw cycles.

Rock fragments within the soil will reduce the effective porosity and thereby lead to faster saturation of the soil. Moreover, rock fragments on the soil surface affect the infiltration rates. When modelling on catchment scale, rock outcrops and areas with impermeable hardpan (e.g. around settlements) have to be considered as such areas may result in increased localized erosion.

Areas susceptible to crusting pose special problems, as the crusting/sealing can cause quick changes of the hydraulic conductivity. From the experiments by Bryan and Poesen (1989) it seems important to operate with separate infiltration rates for interrill areas and within the rills, particularly in areas subject to crusting. Rocks that are embedded in a surface seal reduce infiltration while rocks on the surface protect the soil structure and promote infiltration (Poesen and Ingelmo-Sanchez, 1992; Poesen et al., 1994). An improved description of soil erosion on crusting soils must include a feed-back system between the soil erosion model and the hydrological model - e.g. the effect

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of raindrop detachment on crust development and thereby the hydraulic conductivity which in turn affect the amount of overland flow.

The surface storage capacity influences both infiltration and the time till runoff occurs. It is, among other things, determined by soil type, slope steepness, type and in particular orientation of tillage. Depression storage may be negligible on smooth seedbed tilled up and down the slope, whereas a ridged potato field with the ridges following the contours will have a considerable surface storage depth. Based on laboratory data Morgan et al. (1995) related the surface storage depth, D (mm), to the ratio of the straight line distance between two points on the ground (X) to the actual distance measured over all the micro-topographic irregularities (Y), (see figure 2b):

$$D = \exp(-6.6 + 27(\frac{Y - X}{Y})) \tag{4}$$

This equation does not consider the depths of the depressions and the slope on which they are measured. Although guide values for depression storage may be obtained for various combinations of soil type, slope steepness, tillage methods, and orientation of tillage, the issue of surface storage depression is complicated, e.g. by the fact that roughness element break-down takes place over time.

3.1.3. Soil Surface Conditions and Runoff Processes

Surface flows are influenced by irregularities of micro-topography, caused by management practice, vegetation, and soil clods/aggregates. This results in an uneven distribution of the flow over the surface and influences surface roughness. The process is further complicated because the micro-topography varies considerably during the year, due to management practice, vegetation, etc.

The basic equations for describing surface runoff, Q, as well as sediment discharge, q_s are the conservation of mass equations for flow and sediment, respectively. In the EUROSEM (Morgan et al., 1995) the computation of runoff and sediment is based on a numerical solution of the dynamic mass balance equation (Bennett, 1974; Kirkby, 1980; Woolhiser et al., 1990):

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} = r(t) - i(t) \tag{5}$$

$$\frac{\partial AC}{\partial t} + \frac{\partial QC}{\partial x} - e(x,t) = q_s(x,t) \tag{6}$$

where A is the cross-sectional area of the flow (m^2), Q is the discharge ($m^3\ s^{-1}$), r(t) is the rainfall less the interception for interrill flow and the unit discharge into the rills (from interrill areas) in rill flow ($m^2\ s^{-1}$), i(t) is the local infiltration rate ($m^2\ s^{-1}$), x is the horizontal distance (m), t is the time (s), C is the sediment concentration ($m^3\ m^{-3}$), e is the net detachment rate per unit length of the flow ($m^3\ s^{-1}\ m^{-1}$), and q_s is the

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external input of sediment per unit length of flow (m^3 s^{-1} m^{-1}). For interrill flow q_s becomes zero.

By using the kinematic wave assumption (and that Q = A v) equations (5) and (6) can be solved by using the Manning equation for flow velocity. Although comprehensive guideline values for Manning's roughness coefficient exist (e.g. Engman, 1986), the use of equations (5) and (6) is associated with many difficulties. Due to the variety of surface conditions in the field one may question how representative these guideline values are. Most data available are still based on laboratory experiments, and Emmett (1978) found that field data indicated a ten-fold increase in resistance on the natural field plots compared to the laboratory surfaces. The large temporal variation in surface roughness over the year as a result of tillage, soil consolidation and rainfall further complicates the situation. Savat (1980) found that the Manning equation underestimates the friction coefficient for thin sheet flow.

Most existing attempts to describe surface roughness, e.g. the MIF-index (Römkens and Wang, 1986), only provide qualitative results. No procedure has yet been discovered allowing field measurements of surface roughness to predict accurately hydraulic roughness coefficients independently of flow measurements. The need for such linkage between physical surface roughness and hydraulic roughness is obvious in soil erosion modelling. This necessarily also has to include a method to estimate the temporal changes in soil surface conditions.

Even with proper hydraulic formulas for thin overland flow, the uneven distribution of the water over the soil surface complicates the description. The uneven distribution has a major impact on the transporting and detaching capacity of the flow and thereby the initiation, development, and spatial location of rills, and the distribution between rill and interrill areas.

The two most important factors regarding the flow distribution is the topography and the tillage orientation. On an experimental plot tilled up-and-down, situated on a uniform slope, the flow direction and distribution are relatively easy to predict, but when it comes to small catchments sloping in more than one direction and tilled along the contours or at an angle to the slope, the routing soon becomes complicated. In such case it requires both detailed input data on micro-topography and macro-topography and a model able to route the water in all directions and not only down the prevailing slope direction.

In cases where the area is tilled up and down the slope, a hypsometric curve (Styczen and Nielsen, 1989), showing the frequency distribution of heights over a unit length perpendicular to the flow direction, can be used to quantify the distribution of flow depths in the rills/depressions.

Concentration of water into rills will greatly increase the detachment as well as the transport capacity of the flow and therefore a correct routing of water as rill flow and interrill flow is important. A good correspondence between observed and simulated hydrographs does not automatically mean that the description of overland flow is correct as the routing between rills and interrill areas may not be correct. To obtain this a hydrological model with a detailed description of the overland flow routing is needed (see Section 3.2).

3.1.4. Soil Detachment and Transport by Overland Flow

It is generally agreed that without occurrence of surface runoff, erosion rates are small. Erosion becomes really severe when the overland flow gains enough power to detach the soil, and rill and/or gully erosion occurs.

The detachment and transport of soil by flow are two issues still under discussion. And although interrelated it is necessary to distinguish between the two. The concept of transport capacity is originally developed for non-cohesive materials, meaning that the shear strength of the material is rather low. Most soils, however, are cohesive materials, with a somewhat higher shear strength. In addition, most soils consist, not of single particles, but of more or less water-stable aggregates, which, however, to some extent may break down under influence of water and physical action.

Looking solely at flow detachment, the concentration in the flowing water is determined as a balance between detachment (which is influenced by the mean flow velocity, the shear stress of the water and the shear strength of the soil, as well as the surface contact area), and deposition. As the final concentration is influenced by the shear strength of the soil, it may be different from what it would have been, had the material been noncohesive. However, in addition to the material detached by flow there may be an addition of material through splash, and the two types of material confuse the discussion.

Meyer and Wischmeier (1969) proposed that the detachment capacity at each point should be compared with the transporting capacity and the actual transport rate at that point. The actual net detachment rate is then taken as the lesser:

$$\frac{\partial C}{\partial x} = D_c \text{ if } C < TC, \qquad C = TC \text{ if } C \geq TC \tag{7}$$

TC is the transporting capacity, C is the actual sediment load, and D_c is the detachment capacity. In the alternative approach by Foster and Meyer (1972) the net detachment rate is related to the deficit between the actual sediment load and the transport capacity load:

$$\frac{\partial C}{\partial x} = \frac{TC - C}{h} \tag{8}$$

TC, C and D_c are defined as above, and h may depend on other variables. As the net detachment rate reflects a balance between detachment and deposition processes, the basic approach of equation (8) is used in most physically-based erosion models (e.g. Lane and Nearing, 1989; Morgan et al., 1995). However, without a proper description of the detachment and deposition processes their physical background is still not properly understood.

The most convincing physical description of sediment concentration being a balance between detachment and deposition is made by Torri and Borselli (1991) who base their description on the following assumptions:

(a) hydraulic roughness decreases with increasing sediment load;

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(b) overland flow detachment is proportional to the part of boundary shear stress due to the water fraction of total (water+sediment) fluid discharge;

(c) sediment deposition follows Stokes law of motion (and final velocity of the particles is not reached in the thin flows in question).

The algorithm proved to be a good approximation of the physics behind the empirically derived transport equation by Govers (1990):

$$TC = c(\omega - \omega_{cr})^{\eta} \tag{9}$$

where S is the slope, ω is the stream power (u S). u is the mean flow velocity (cm s^{-1}). S is the slope, ω_{cr} is the critical value of unit stream power (= 0.4 cm s^{-1}) and c and η are experimentally derived coefficients depending on particle size. The algorithm derived by Torri and Borselli (1991) is also able to reproduce the empirical equations for incipient rilling which link shear velocity with soil shear strength (e.g. Rauws and Govers, 1988). The theoretical considerations by Torri & Borselli (1991) aiming at combining physical descriptions and empirical results seem to form a good basis for improving the understanding of the interactions between flow detachment and transport processes of thin overland flow.

Until the recent development of transport capacity equations for thin overland flow a variety of "classical" transport capacity equations for streamflow have been used including the Engelund-Hansen (Engelund and Hansen, 1967) and the Yalin (Yalin, 1963) equations.

Following the logic of the balance approach, material added through splash will increase the concentration in the flow, but the material will deposit following the same rules as the flow-detached material. Models using the approach of eq. 7 have a tendency to deposit material in lumps rather than gradually because the sediment concentration is supposed not to exceed the transport capacity determined for non-cohesive materials. However, the transport capacity is determined from flow detachment alone and does not take into account that material may be added through other processes, without energy expenditure of the flow.

Settling velocities should be calculated separately for groups of particles or aggregate sizes. The use of one d_{50}-value may cause instabilities during modelling as decreasing transport capacity may result in a very abrupt increase in sedimentation rates. Some of the questionable issues may still be calculation of settling velocities using Stoke's law in thin overland flow with turbulence caused by irregularities, and how to treat flows and sediment transport for aggregates that may have a mean diameter larger than the flow depth. Torri and Borselli (1991) calculate effective shear stress as a function of the hydraulic radius minus the sediment diameter, and thus the equations only apply for water depths larger than the sediment diameter.

According to Torri and Borselli (1991), the soil particles are kept in place by the cohesive forces and the submersed weight of the particles. In some equations this is described by the detachment rate being reduced by a coefficient ß, which decreases with increasing shear strength, τ_s.

However, the soil shear strength varies through the year as a result of changes in moisture content, tillage operations, impact of plant roots, age hardening, etc. In

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addition it changes very fast when the soil is wetted. Table 2 clearly illustrate the dynamic nature of the shear strength due to wetting and drying. Effects of tillage, roots etc. will further complicate the picture. The relative changes in soil shear strength during a rainstorm will among other things be influenced by initial soil moisture content and soil type. Sandy soils will show a larger change upon wetting as a relatively large part of their cohesiveness prior to the rain is due to suction (Andersen & Lørup, 1991) (Table 2). Govers et al. (1990) found that shear strength measured on a saturated soil correlates best with erodibility.

The lack of a physical description of the shear strength and its variation is one of the major constraints in soil erosion modelling presently. By using a continuous model it may eventually be possible to simulate the variation of the shear strength by relating it to the changes in soil moisture, root development, time after tillage, etc., but this will require a better understanding of the dynamic nature of the shear strength. In the longer term, model developers will have to rely on continuous models to predict some of these crucial changes of parameter values. Simulations carried out by Wicks et al. (1992) does indicate that is the way forward.

TABLE 2. Measured values of shear strength for 2 different soils with different content of soil organic matter (SOM) as a function of time since overland flow ceased (Andersen & Lørup, 1991).

Soil type	Shear strength (kPa) after 15 minutes with overland flow	Shear strength (kPa) 16 hours after overland flow has ceased	Shear strength (kPa) 7 days after overland flow has ceased
Loamy sand (1.34% SOM)	0.20	1.55	2.49
Sandy loam (2.34% SOM)	1.10	1.95	2.82
Sandy loam (2.34% SOM) sieved (< 2 mm)	0.50	1.52	2.10

Modelling of transport capacity on cohesive soils is still subject to uncertainty as most work has been carried out using non-cohesive material and the conclusions may not hold for cohesive soils. Here the major part of the soil mass is aggregates having dry (and wet) densities below 2 g cm^{-3} and mean diameters many times higher than d_{50} of the soil mass. A decrease in wet bulk density from 2.65 to 2 g cm^{-3} implies that the Shield parameter, θ, will increase 60%.

3.1.5. Description of Rill Initiation and Development

In principle, flow detachment and sediment transport in rills follow the same rules as discussed in subsection 3.1.4. However, due to the concentration of flow, the final erosion rates are generally much greater.

The processes which need special concern are rill initiation, headcut retreat, wall collapse, and rill tail development.

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Rills develop when the shear stress of the flow is large enough to remove "all" sizes of soil particles at particular spots along the slope. Rauws and Govers (1988) used the empirical equation:

$$U_{gcr} = 0.89 + 0.56C \tag{10}$$

where U_{gcr} is the critical grain shear velocity of the flow (cm/s) and C is apparent cohesion (kPa). Other approaches (e.g. Torri et al., 1987a) suggest τ_0/τ_s to exceed a certain constant value, τ_0 being the flow shear stress and τ_s being the soil shear strength of the soil top layer. In cases where vegetation is present, it may be better to use $\tau_0{*}v/\tau_s$ as an indicator, because a larger roughness gives rise to larger water depth, but only a certain part of the resulting stress actually acts on the surface particles.

Headcut retreat has been suggested described as a function of the potential energy released during the drop of the water into the headcut (Styczen and Nielsen, 1989) or being proportional to the total mass of the water at a given time multiplied by the mean flow velocity before the drop and inversely proportional to the stability of the headcut walls (De Ploey, 1989).

A certain part of the shear stress of the flow in rills will act on the walls of the rill. This will cause erosion. If undermined, the upper part of the rill walls is likely to collapse and provide easily detachable material to the flow. A laboratory study on rill initiation and development (Andersen & Lørup, 1991) showed that on sandy soil where the inherent cohesion is low the rill walls easily collapsed and the rills tended to be shallow and wide, whereas rills on clayey soils were deeper but relatively narrow.

The position of the rill tail is determined by the slope of the rill bottom compared to the slope of the hill (Styczen & Nielsen, 1989). As long as the bottom of the rill is flatter than the hill slope, the rill will continue to develop downslope because the transport capacity of the water is larger outside the rill than inside, and this causes continuous erosion. When the two angles/slopes become equal, downward development ceases.

From a rill model based on the principles described above (DHI and IoG, 1992), the following observations were made. The end point of the rill was easily defined, and when reached, it was independent of simulation time. The width of the rill was not independent of time, but the development slowed down as the rill grew wider, because the depth of flow decreased, and exerted less stress on the walls. The rill depth was dependent on the speed of headcut retreat (and wall collapse), because this determined the rate of net detachment from the bottom. The final shape of the simulated rill bore close resemblance to the rill from which input data were generated.

Presently, most models use predefined shallow rills/depressions rather than allow them to be initiated during the simulation. Particularly on agricultural fields this may be defendable, as the flow pattern to a large degree is determined by the tillage operation, and both spacing and direction can be described. However, the critical issue is whether the hydrological model manages to describe the flow pattern and the confluence of water at certain points in the field, triggering incision. It may be attempted to describe rill initiation through the use of a hypsometric curve or a

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statistical evaluation of probability of rill occurrence, based on the distribution of shear stress and shear strength or similar parameters.

3.2. COUPLING WITH HYDROLOGICAL MODELS

During the early days of developing soil erosion prediction models these models were developed parallel to and independent of hydrological models, and there was little collaboration between agronomist/agricultural engineers and hydrologists. This was mainly due to the fact that the first models were purely empirical and did not need input from hydrological models. Moreover - in particular among agronomists/agricultural engineers - soil erosion was originally considered mainly as a problem in relation to agricultural production. However, in the 1970s the development of conceptual models and the attention to erosion as a potential cause of pollution of water bodies raised the need for hydrological models to be coupled to erosion models and for a closer collaboration between various disciplines of science. With the present development towards physically-based erosion models collaboration and appropriate hydrological models have become even more crucial.

One of the main aims of a physically-based soil erosion model is to describe the various processes as they appear in the natural system. This necessarily requires that such a model is coupled to a hydrological model that fulfil the same aims, i.e. a model which can provide a detailed description of the spatial and temporal changes in the flow of water. A good agreement between simulated and observed amounts of soil loss/ sediment yield does not indicate model-predictive credibility without a similar good agreement between simulated and observed discharge hydrographs.

Due to the need for a detailed description of the overland flow and the need to simulate the geomorphology of the developed rill the model must be able to work with small grids/elements, preferably down to a few metres. The use of small grids/elements requires the model to run with very small time steps.

Furthermore the hydrological model should be a continuous model with a so-called "hot start" facility. This implies that the model can be used as an event model on the basis of initial conditions retrieved from previous model runs. Prior to the "hot starts" the hydrological model may be run for the whole period of concern. Hereby it is possible to identify the major rainfall events where significant overland flow has been generated and soil erosion is likely to have taken place. Furthermore the result file from this run will provide important input variables, such as initial moisture contents, for running the coupled hydrological and soil erosion models for specific storm events - data which otherwise would have to be collected in the field prior to the rainfall events.

As soil erosion modelling calls for a more detailed description of overland flow than hydrological modelling normally does, the use of existing hydrological models for soil erosion modelling purposes may therefore require a revision of the overland flow component.

The separate modelling of rill and interrill erosion requires separate routing of interrill and rill (concentrated) flow including routing of water from interrill areas to the rills. The hydrological model must include a two-dimensional surface description

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where tillage orientation may be across the slope or at an angle to the slope and not just downslope. Otherwise this will restrict the application of the model seriously.

The distinction between interrill and rill flow and the way they interact are certainly some of the major challenges in the development of hydrological models to be used in soil erosion modelling.

4. Construction, Calibration and Validation of Soil Erosion Models

4.1. GENERAL CONSIDERATIONS

Soil erosion modelling is a complicated issue, which requires a solid understanding of the different hydrological and soil erosion processes in general and specific data for the particular study area. As a guideline for the various steps to consider and a consistent terminology to use the modelling protocol outlined by Refsgaard (Chapter 2) may be applied.

The first step to consider is the definition of the purpose of the model application. For physically-based erosion models two groups of users can - broadly speaking - be identified (Quinton, 1994): 1) Field personnel and policy makers, and 2) Researchers. The different types of purposes of a model application may be described as follows:

(a) To test alternative theoretical process descriptions and in this way improve the physical understanding (researchers).

(b) To test the range of model validity in terms of scale of application (erosion plot, hillslope, catchment), conditions on soil and geology, land use and hydroclimatological regime. This is very important in order to avoid misuse by extrapolating the use of the model beyond its proven range of applicability (both researchers and field personnel/policy makers).

(c) To predict soil losses/sediment yields and the effects of possible management options regarding soil and water conservation measures at particular localities (field personnel/policy makers).

4.2. ESTABLISHMENT OF A CONCEPTUAL MODEL AND SELECTION OF MODEL CODE

The next step is to establish a conceptual model - the modelling framework. This includes identification of the key processes required to be included in the model. The processes to be included and the degree of details to which they need to be modelled depend on assessments of the relative importance of the various processes in the particular study area, the acceptable accuracy limits, the data availability and the specific purpose of the study. At this stage it is important through readily available information to test ('model qualification') whether the defined conceptual model in qualitative terms appears to be a good representation of the physical system.

On the basis of the conceptual model an appropriate model code must be selected. In this connection it is important to select a code that can describe the processes included in the conceptual model. For instance, a model code that is only able to

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describe Hortonian type of overland flow should be avoided, if the study areas is characterized by significant saturated overland flow; or, similarly, if a model code is unable to describe rill erosion, it should not be applied to an area, where rill erosion is the prevailing type of erosion. Usually, an appropriate code will be selected among existing codes. However, in connection with research projects development of new or modified codes may be important. In such cases the new codes need to be verified, i.e. tested for their ability to provide mathematically accurate approximations to the given process equations.

4.3. MODEL CONSTRUCTION

Model construction involves data collection and processing into appropriate model formats, assessment of parameter values, definitions of model boundaries and internal discretizations. One of the major difficulties in the application of distributed physically-based soil erosion models is the large data requirements. As the ideal requirements of measured field data for all parameters for all model grid points never will be available, the main part of the parameter assessment will have to be done from secondary information, such as mapped soil and vegetation types. The spatial and temporal variability of model input is one of the major challenges to physically-based distributed erosion models. Thus the *parameterization*, i.e. the proces of defining structures of parameter variations, is a crucial part of the model construction. The spatial variation in parameter values should as far as possible reflect the variation found in the field and the availability of data. As vegetation characteristics and tillage operations are major factors in relation to soil erosion, the spatial variation may depend on the number of fields and the topography, unless there are major changes in soil type within the fields. Similarly, information on temporal variation of certain model parameters, such as annual variation of vegetation parameter values, could be incorporated in the parameterization. For continuous models the temporal variability of certain parameters may partly be measured and partly included in the model simulations, as discussed in section 3.1.4. This may reduce the data input requirement considerably.

4.4. MODEL CALIBRATION AND VALIDATION

The first stage in the calibration is calibration of the hydrological model, which in most cases implies calibration of simulated and observed hydrographs. This is followed by calibration of simulated and observed sediment rates. In cases where the insight into the functioning of the erosion component is the main aim, this should preferably be done when good agreement between hydrographs has been obtained - otherwise the calibration of simulated and observed sediment rates may deteriorate the erosion component rather than improve it.

Even if a reasonably good agreement is established between calibrated simulations and observed data, and the parameter values seem physically reasonable, it is not until the model has been validated for a number of rainfall events that a picture emerges of the model's ability to simulate the physical environment. During a validation, the model is used to simulate other events than those used during the calibration phase. Only if

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the results of the blind simulation and the observed data resemble each other adequately (according to pre-described performance criteria), the model can be considered validated for that particular situation.

For two-dimensional models as the EUROSEM/MIKE SHE that are able to simulate the spatial distribution of the amount of erosion/sedimentation, rill depths, depths of overland and rill flow, etc., qualitative or semi-quantitative comparison with field observations of location and depth of rills, the amount of sedimentation at specific sites, etc. will be valuable to evaluate the performance of the model.

As resuspension, bed and bank erosion can account for a significant part of the sediment leaving small catchments (Hasholt and Styczen, 1993) this should be considered when calibrating and validating on catchment level.

5. Case study: Application of the EUROSEM/MIKE SHE Soil Erosion Model

5.1. THE MODEL, STUDY AREA AND MODEL CONSTRUCTION

EUROSEM/MIKE SHE is the linkage of the codes of the EUROSEM soil erosion model (Morgan et al., 1991) and the MIKE SHE (Refsgaard and Storm, 1995) distributed physically-based hydrological model. Linking the two models required a new overland flow component to MIKE SHE to make separate routing of rill and interrill flow possible.

The EUROSEM/MIKE SHE provides a comprehensive and detailed description of the hydrological processes including generation of saturated overland flow. In contrast to the EUROSEM/KINEROS code the MIKE SHE version gives a continuous simulation of e.g. soil moisture content, and the rills may run at an angle to the slope and not necessarily perpendicular to the contour lines.

The EUROSEM/MIKE SHE model was tested on a 25 x 35 m^2 soil erosion plot at Woburn Experimental Farm, UK (DHI, 1994). The site has been subject to erosion since at least 1950 (Catt, 1992). The plot is non-uniform with a mean slope of 9% in the main sloping direction (see Fig. 4). The soil is mainly the Cottenham series, a dark brown loamy sand. The plot was tilled up and down the slope and sown with sugar beets approximately a month before the studied rainfall events.

The plot was divided into 2.5 x 2.5 m grids with 2 rills (shallow depressions as a result of tillage) pr. metre width in the tillage direction. The vertical discretization in the unsaturated zone was 2 cm for the upper 10 cm increasing gradually to 0.5 m below 2 m depth. Rainfall data were available on a daily basis prior to the storm and on a minute basis during the main part of the storms. Daily values for evaporation were used. The model was parameterized according to EUROSEM input files for the storms.

5.2. CALIBRATION AND VALIDATION OF EUROSEM/MIKE SHE

Two rainfall events on May 29, 1992 were included in the test. The first event was used to calibrate the model, after which the model was validated on the second event using the parameter values obtained during the calibration of the first event.

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Only a few of the original parameters in the EUROSEM input file were changed during calibration of the hydrographs, mainly roughness coefficients, saturated hydraulic conductivity, surface detention storage and cohesion. As no information on the groundwater level was available the depth to groundwater was used as one of the main calibration parameters to fit the simulated and observed hydrographs. All the parameter values were kept within physically realistic limits. For example, the value used for saturated hydraulic conductivity (12 mm/h) corresponded well with the mean of the measured values (12.01 mm/h). For comparison, the value used in EUROSEM/KINEROS simulation of the storm was 2.0 mm/h (Quinton, 1993).

TABLE 3. Comparison between simulated and observed values for runoff and soil loss from a 25 x 35 m plot at Woburn Experimental Farm for two rainfall events on May 29, 1992. The first rainfall event (29.5a) was used for calibration, and the parameter values were then used to run the model for the second event (29.5b).

Rainfall events in 1992	Rainfall		Runoff		Soil loss	
	Total rainfall (mm)	Max. intensity (mm/h)	Observed (m^3)	Simulated (m^3)	Observed (kg)	Simulated (kg)
29.5a	3.19	40.8	0.576	0.623	4.16	4.13
29.5b	4.12	26.4	0.208	0.269	1.68	1.40

Simulation results for the calibration event are shown in Fig. 3. The volume of the observed and the simulated hydrographs was reasonably well matched, whereas it was difficult exactly to match the shape of the observed hydrograph, especially the beginning of the hydrograph. This may be due to uneven distribution of surface detention storage over the surface and/or slightly wrong initial soil moisture content.

The matching of the simulated and observed sediment rates resembles that of the hydrographs; while it was difficult to match the exact shape of the curves (Fig. 3), the simulated value for the total soil loss corresponded well with the observed value. Using the parameter value in the EUROSEM input file (except those modified during the calibration of the hydrograph) resulted in a considerable overestimation of the sediment transport. As the majority of the soil loss is caused by rill erosion, the mean diameter and cohesion were used as the main calibration parameters.

It should be noted that for both rainfall events the values for the total runoff and the total soil loss are very small, as the total soil loss from the two events is 0.07 t/ha. Thus, even a small *absolute* error in the simulated values will result in a large *percentage* error. In general, the use of major rainfall events with considerable runoff and erosion would give a better possibility to test the performance of soil erosion models.

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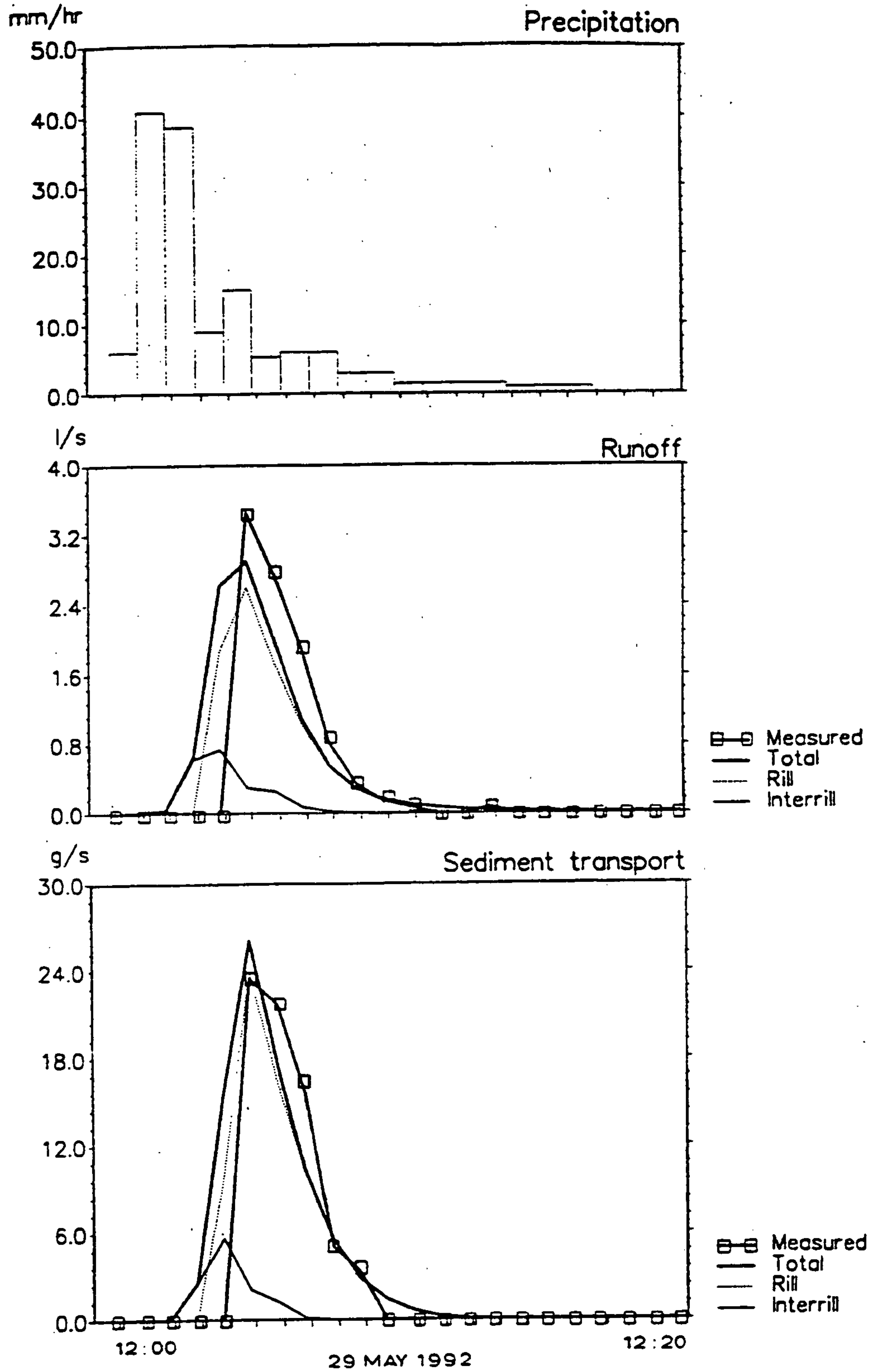

Figure 3. Rainfall, simulated and observed hydrographs, and simulated and observed sediment rates for the first rainfall event on May 29, 1992 on a 25 x 35 m^2 erosion plot at Woburn Experimental Farm, UK.

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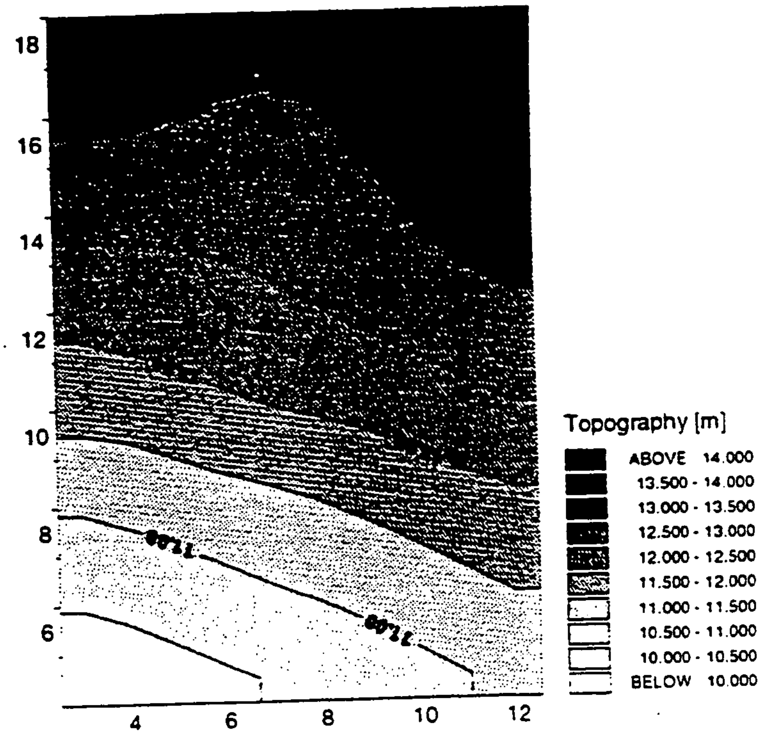

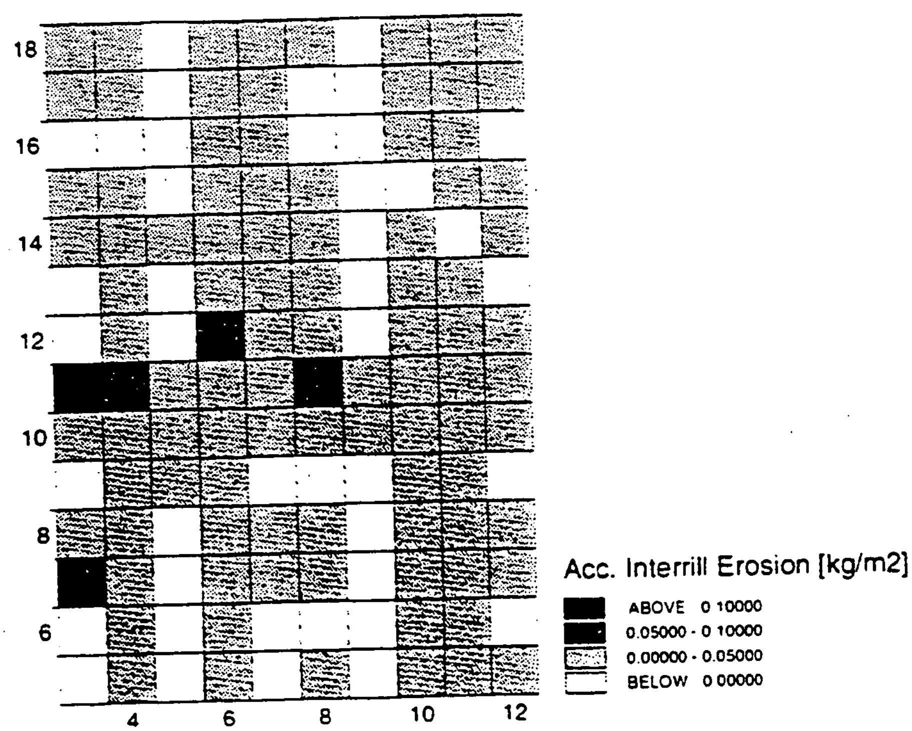

Figure 4. Topography and model grid for the Woburn erosion plot (upper figure) and two-dimensional representation of interrill erosion and sedimentation for the first rainfall event on May 29, 1992 (lower figure). Positive values imply erosion, while negative values correspond to areas with sedimentation.

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Some of the spatially distributed features of the model are illustrated in Fig. 4. The topography is shown in the upper part of Fig. 4, and the accumulated simulated interrill erosion and sedimentation are shown in the lower part of the figure for the first rainfall event on 29 May 1992 (same event as shown in Fig. 3).

From Fig. 4 a very significant spatial variability is noticed. Thus, the net erosion rates shown in Fig. 3 turn up to be a result of erosion taking place over the main part of the area minus sedimentation over smaller areas. By comparing the erosion maps with the topography in Fig. 4 it appears that the small topographical 'irregularities' generate differences in overland flows (not shown here) which again generate significant spatial variations in the erosion pattern.

5.3. SENSITIVITY ANALYSES

Although a rather good prediction of runoff as well as of soil loss was obtained, it became obvious during the calibration that especially the hydrological part of the model was very sensitive to changes in parameter values. Thus, for the first storm (29.5a) a number of sensitivity tests was carried out for the most sensitive parameters (Table 4).

The extremely high sensitivity to the hydraulic conductivity is mainly due to the fact that the rainfall intensity for a major part of the storm was in the same order of magnitude as the hydraulic conductivity. For storms with large differences between the rainfall intensity and the hydraulic conductivity the model will be less sensitive to the hydraulic conductivity. The effect of the number of rills shows the effect of the concentration of the overland flow and illustrates the importance of a good surface description. The effect of splash erosion will vary from storm to storm depending on the amount of runoff generation during the rainfall event.

TABLE 4. Sensitivity analyses of EUROSEM/MIKE SHE on a 25 m x 35 m plot at Woburn, UK on May 29, 1992 (DHI, 1994). The observed runoff and soil loss values were 0.576 m^3 and 4.16 kg.

Parameter	Parameter change		Simulated runoff		Simulated soil loss	
	From... to	%	m^3	Change %	kg	Change %
Original parameters			0.622	-	4.16	-
Manning n ($m^{1/3}$ s)	0.033->0.028	- 21	0.639	+ 3	4.93	+19.4
Surface detention storage (m)	0.0004->0.0005	+ 20	0.529	- 15	3.22	- 22
Saturated hydraulic conductivity (mm/hr)	12->11	- 8	0.692	+ 11	4.68	+ 13
Splash erosion included	Yes->No		0.622	0	3.64	- 11.9
Number of rills per meter width	2->1		0.678	+ 9	4.92	+ 19
	2->3		0.566	- 9	3.58	- 13

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6. Discussion on Limitation, Applicability and Research Needs

While the last decade has seen substantial progress in the process descriptions for soil erosion, and considerably increased the understanding of the interaction between the different processes, there are still basic issues that complicate the modelling.

Sediment transport is highly dependent on the pattern of overland flow. Outside the laboratories, description of flow patterns, roughness elements, flow velocities, and the division of shear stresses between the surface soil and other roughness elements have turned out to be very difficult. Spatial differences in infiltration also play an important role.

An often mentioned limitation of the physically-based distributed models is the amount of data needed to run the models. The issue is dual, because the detailed process descriptions and the extensive data collections have generated much more detailed knowledge about how, where, and under which conditions erosion takes place. It is true at present that the data requirements appear comprehensive, but as more understanding of sensitivity in different environments is generated, it also becomes possible to target the data collection to a higher degree than is presently done.

Thus, a direct application of soil erosion models on medium to large size catchment scales is not yet feasible in practice. The empirical models can be used together with GIS's to prepare qualitative information such as erodibility indexes, but they can not be expected to provide reliable quantitative predictions, and they are not well suited to assess the impacts of alternative soil conservation management options. The physically-based models, on the other hand, are not yet ready for application at such scale. This does not imply, however, that soil erosion models are not useful tools for soil and water management. When used with great care by experienced modellers model results may be very useful for practical purposes.

Examples of possible approaches for applications of distributed physically-based models at different scales include the following:

(a) *Plot and hill slope scale*. Here it is feasible directly to apply a physically-based soil erosion model.

(b) *Small size catchment* (up to a few km^2). Here it is possible to use distributed physically-based models, but maybe some of the process descriptions requiring the finest spatial and temporal resolution, such as rill erosion, may have to be described rather coarsely. Such simplification implies that the models may not be able to simulate all key processes to the same degree of reliability; however, still the results may be very useful.

(c) *Ordinary size catchment* (up to several hundreds or thousands of km^2). In this case there are different ways of utilizing model results, such as:

- If model parameters for a particular typical site are known, it may be possible - for given rainfall events - to tabulate runoff rates and sediment rates as a function of different interventions. Such tables could aid extension workers in choosing conservation methods according to the sensitivity of that particular type of site. It should be noticed that the traditional empirical

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models do not allow extension workers or farmers to judge, for instance, whether, at a particular site, infiltration is the most efficient parameter to manipulate, or whether surface runoff cannot be avoided, so the conservation methods therefore must focus on removing excess water. Potential applications of a physically-based soil erosion model (EUROSEM) for evaluating the effects of soil conservation measures is reviewed by Rickson (1994).

* Soil erosion modelling is carried out at on a number of representative plots or hillslopes, so-called 'soil erosion response units'. The entire catchment is then divided into sub-units and each sub-unit classified as being represented by one of the erosion response units. These response units are characterized by common slope, soil type, land use, climatic regime and possibly other factors. This approach is very suitable for a combination of a soil erosion model and a GIS.

Obviously, when moving from modelling at plot scale to modelling at catchment scale the model accuracy becomes less at each point in the area; but experience indicates that the integrated output from the catchment is not necessarily much less accurate than the model output from the plot scale.

Some of the critical issues which need to be addressed in future research include:

(a) The variation of soil shear strength over the year as well as during a rainstorm as a function of soil moisture content, tillage, vegetation and time, so that such changes can be modelled in continuous soil erosion models.
(b) Relations between soil surface roughness (including effects of vegetation) and the hydraulic roughness of the flow.
(c) Transport capacity of thin overland flow where soil material mainly is transported as aggregates rather than single grains.
(d) Interaction between soil and vegetation properties which influence the hydrological processes and parameters, such as hydraulic conductivity.
(e) Effects of tillage on parameter values.

At present, the physically-based soil erosion codes are not much applied outside the group of researchers who have been involved in the development of the codes. A main reason for this, in addition to the problems outlined above, is that these codes are generally not very well documented and not very user friendly. The necessary technological innovations in this regard can be expected to be made gradually, as the research results improve the model applicabilities and as the demands for model use increase.

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CHAPTER 7
AGROCHEMICAL MODELLING

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1. Introduction

1.1. BACKGROUND

During the last decade problems with increased emission of agrochemicals have become more and more obvious. In a number of countries, the stage is now reached, where political decisions regarding control of such emissions have been taken or in the process of being taken and alleviation measures are being implemented. Most of these measures are based on rough estimates of the risk of agrochemical pollution, which not always consider the interaction between climate, crop, soil and hydrology, and additional tools for assessment of the long term effects of the suggested measures of pollution control are lacking. It is therefore very relevant to investigate which tools are available for prediction, how reliable they are, and what are their limitations.

Intensive large scale monitoring programmes are being established in many countries aiming to assess the magnitude of the pollution problems from non-point sources in rural areas. However, such programmes are only able to identify the problems and quantify the results. They are not capable of performing cause and effect analysis in order to identify critical areas with high pollution risk or critical management practices causing higher losses than others. For this purpose understanding of the processes responsible for transport and transformation of the chemicals in the various hydrological compartments is crucial. In this respect, mathematical models describing the relevant processes provide strong tools which can support the interpretation of the monitoring results and provide the possibility to investigate and compare the effect of different management practices on potential losses to the surrounding environment.

1.2. TYPES OF MODELS AVAILABLE

A large number of computer codes have been developed in the past years to describe the transport and fate of agrochemicals in the different parts of the hydrological cycle. They vary in complexity, ranging from simple empirical formulas to comprehensive distributed physically/chemically-based descriptions. Traditionally, there has been a distinction between leaching models and field or catchment models. Leaching models are confined to one dimensional descriptions of the root zone processes, while field or catchment models consider smaller or larger parts of other surface and subsurface processes.

M. B. Abbott and J. C. Refsgaard (eds.), Distributed Hydrological Modelling, 121–141.

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It is outside the scope of this chapter to give a comprehensive review of existing model codes. The main emphasis will be put on presenting a few state-of-the-art descriptions for some of the most promising tools, both from a research and a management point of view, and to describe the principal differences in modelling methodologies.

2. Process Modelling at Point Scale

2.1. GENERAL

A large number of model codes describes the unsaturated zone, including the root zone, in a single profile. The important leaching models are deterministic models, implying that they, as far as possible, describe the physics and chemistry of the processes. The following sections briefly present state-of-the-art leaching model codes describing transport and transformation of nitrogen, phosphorous and pesticides.

When evaluating the features available in leaching models, some general considerations regarding the process requirements are necessary. A prerequisite for describing solute movement in soils is that the description of water flow and the available boundary conditions are adequate for the situation under consideration. The models must be able to handle the hydrological conditions present in the soil. For instance, if shallow groundwater is present, the selected model must be able to handle groundwater fluctuations and capillary rise. Some common approaches and related assumptions are described in Table 1.

Another important part of the water balance which must be considered is the evapotranspiration. Several different approaches exist implying that the actual climatic conditions must be analysed when evaluating the simulated water balances. Additionally, the chemical transformation processes included should reflect the current knowledge regarding processes having significant influence on solute behaviour in the soil.

2.2. NITROGEN MODELLING CODES

The impacts of agricultural crop production on the environment in terms of nitrogen losses to surface water and groundwater is related to the input level of fertilizers as well as the structure of the cropping system. It is well known that eg high application rates of organic manure, and in particular cropping systems without crop cover during periods in which mineral nitrogen is released from organic matter in the soil, may result in increased nitrogen losses in subsequent periods with water discharge. During the recent four decades nitrogen losses from rural areas to the aquatic environment have increased causing deterioration of the water quality and subsequently created great concern on how environmentally and economically sustained agricultural crop production can be developed.

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TABLE 1. Examples of approaches and assumptions used in leaching models.

Process	Approach	Assumption
Water flow	Capacity model	Water flow depends on the storage capacity of each layer + an empi cal drainage rule. Flow between layers only occurs when the capacit is exceeded. Capillary rise is not taken into account.
	Richards' equation	Water flow depends on the hydraulic gradient and the soil physical properties (hydraulic conductivity and soil water retention curves), a is calculated dynamically for the entire column. Capillary rise is automatically accounted for.
	Preferential flow paths considered	Soil matrix contains macropores or similar preferential flow paths which, when activated, transport water at fast rate from surface laye towards the bottom of the root zone.
Solute transport	Piston displacement	Convective transport with water flow only. Dispersion is set by the user or indirectly accounted for by numerical dispersion.
	Convection-dispersion equation	Convective and dispersive transport assumed. Hydrodynamic dispersion calculated.
	Mobile/immobile water considered	Soil matrix divided in active, mobile fraction where the water movement takes place and an immobile fraction. Diffusion between mobile and immobile phases.

Agricultural crop production as well as losses of nitrogen is determined by a number of physical, chemical and biological processes in the soil-plant-atmosphere continuum which interact simultaneously in a complex way. In the nitrogen cycle in the soil-plant system, the pathway of nitrogen is a complex series of transformation and transport processes all of which are affected by external factors. Thus, it is difficult to predict how changed management practices will effect crop production, nitrogen use efficiency and nitrogen losses. In the conventional scientific approach, field experiments have been used to explore possibilities for appropriate system management practices. This type of research has limitations due to the complexity of the system. Simulation models are therefore increasingly used to support experimental research and, though still at minor scale, to assess the effect of legislation meassures.

A large number of model codes exist aiming to describe the interrelationships between energy, water, carbon and nitrogen cycles of the soil-plant-atmosphere system under various external conditions with different levels of complexity.

14 nitrogen leaching model codes were reviewed by de Willigen (1991). Intercomparative tests of five codes were carried out under the auspices of the CEC (CEC, 1991), Hansen (1992) tested and compared two nitrogen leaching modelling codes and Diekkrüger et al. (1995) compared the simulation results of 19 agroecosystem models of which 8 contained approaches related to the nitrogen cycle. Examples of such model codes are reviewed in Table 2.

Table 2. Review of some of the most comprehensive nitrogen modelling codes.

Process	ANIMO	SOIL-N	DAISY	WAVE	RZWQM	LEACHM
Water flow	External model	External model	Richards' eqn.	Richards' eqn. in matrix	Green & Ampt/Richards' eqn.	Richards' eqn.
Preferential flow	-	-	- (+)	+	+	- (+)
Solute transport	Piston displacement	Piston displacement	Conv./disp.	Conv./disp., mobile/immobile water	Piston displacement + mobile/immobile water	Conv./disp.
Boundary conditions	External model	External model	Free drainage, fixed or fluctuating gwt.	Free drainage, gwt., lysimeter, fluxes, pressure head	Free drainage, pressure head	Free drainage, fixed gwt, zero flux, lysimeter
Mineralization	4 OM pools. 1. order kinetics. f(OM pools, T, θ, pH, C/N, O_2-demand)	3 OM pools. 1. order kinetics. f(OM pools, C/N, T, θ)	4 OM pools + 2 BM pools. 1. order kinetics. f(T, θ, Cl, C/N, OM+BM pools, $[NH_4^+ + NO_3^-]$)	3 OM pools. 1. order kinetics. f(OM pools, C/N, T, θ)	5 OM pools + 3 BM pools. 1. order kinetics. f(T, pH, θ, BM pool, I)	3 OM pools. 1. order kinetics. f(OM pools, C/N, T, θ)
Nitrification	1. order kinetics. f($[NH_4^+]$, aeration, θ, T, pH)	1. order kinetics. f($[NH_4^+]$, θ, T, NH_4^+/NO_3^--ratio)	Michaelis-Menten kinetics. f($[NO_3^-]$,T, θ)	1. order kinetics. f($[NH_4^+]$,θ, T)	Zero order at low conc. 1. order at high conc. f(T, θ, pH, BM pool, I)	1. order kinetics. f($[NH_4^+]$, θ, T, NH_4^+/NO_3^--ratio)
Denitrification	Zero order reaction. f(air content, T, θ, O_2-demand, pH, $[NO_3^-]$)	Michaelis Menten kinetics. f($[NO_3^-]$, T, θ, depth)	Denitrific. capacity concept based on CO_2-evol. f($[NO_3^-]$, T, θ)	1. order kinetics. f(T, θ)	1. order kinetics. f(T, θ, pH, BM pool, I)	Michaelis-Menten kinetics. f($[NO_3^-]$, θ)
NH_4^+-adsorption	Linear isotherm	Immobile	Non-linear isotherm	Linear isotherm, different partitioning mobile/immobile sites	Linear isotherm	Linear isotherm
Crop uptake	$NH_4^+ + NO_3^-$ uptake. f(transpiration, flux, $[NO_3^-]$, $[NH_4^+]$)	Predefined or separate module	$NH_4^+ + NO_3^-$ uptake. From calc. pot. N-demand + availability. f(root density, $[NH_4^+]$, $[NO_3^-]$)	From predefined or calculated pot. uptake. f(root density, $[NH_4^+]$, $[NO_3^-]$, diffusive + convective flux)	Pot. crop demand + N-availability. Passive or active transport to root.	2 options. From pot. N-demand + availability or f(root density, $[NH_4^+]$, $[NO_3^-]$)
Volatilization	Given fract. of $[NH_4^+]$ in manure	-	Given fraction of $[NH_4^+]$ in manure	1. order kinetics. f($[NH_4^+]$)	+	1. order reaction
Crop production	-	Separate module	Calc. of LAI, CAI, Dry matter produc. from Rn.	Input or calc. of LAI and Dry matter produc. from Rn.	Calc. of LAI and Dry matter production from Rn.	Crop cover calc. from empirical functions.
Crop modules	-	-	13 crops	7 crops	2 crops	-
References	Berghuijs et al. (1985) Rijtema et al. (1991)	Johnson et al. (1987) Jansson et al. (1991)	Hansen et al. (1990) Hansen et al. (1993)	Vereecken et al. (1991) Vanclooster et al. (1994, 1995)	DeCoursey et al. (1989, 1992)	Wagenet and Hutson (1987), Hutson and Wagenet (1992)

(+): Planned in future versions, OM: Organic matter, BM: Biomass, C: Carbon content, Cl: Clay content, θ: Soil moisture content, T: Temperature, gwt: Groundwater table, I: Ion strength, LAI: Leaf area index, CAI: Crop area index, Rn: Net radiation.

Two of these codes, ANIMO and SOIL-N, only contain descriptions of the nitrogen cycle and hence require an external water flow model. For the codes containing internal flow descriptions two types of approaches are recognized. The more physically-based codes use Richards' equation, whereas the simpler codes use the capacity approach. The basic difference between the two approaches is that capacity models are not able to calculate fluxes based on pressure head gradients. These models are therefore not suitable for conditions with capillary rise. Preferential flow processes described by dual conductivity and/or mobile/immobile water approaches, are only considered in WAVE and RZWQM.

Solute transport is either described by the convection-dispersion equation, corresponding in complexity to the Richard's equation for water flow, or solely by convective transport calculated by multiplying water flux and solute concentration.

All models describe the nitrate transport and transformation in two steps, by first carrying out the water flow calculations, then the N calculations. This approach may be acceptable where N is not a limiting factor, but in situations with serious N deficiency, it may pose a problem. The plant growth simulated in the first step may be optimal from a water availability point of view, but restricted by N deficiency during the second run. The actual evapotranspiration calculated in step 1 will therefore be overestimated, resulting in unreliable estimates of nitrate concentrations and fluxes.

The major differences between the existing nitrogen model codes arise from the approaches applied for describing the components of the N-balance. Especially the complexity regarding mineralization, nitrification, denitrification and plant uptake varies among the models. As these four processes are of major importance for the overall performance of the models with respect to nitrate leaching, the model review presented in Table 2 focuses on differences and assumptions related to the N-dynamics.

All the models describe the kinetics of the mineralization as a 1. order process, but the number of interacting organic pools range from 3 to 7. Only ANIMO, DAISY and RZWQM take explicit account of one or more pools of biomass.

One of the major processes removing nitrogen from the soil profile is plant uptake. Crop N-uptake may be either simulated directly by a crop module accounting for gross photosynthesis, respiration, dry matter and nitrogen distribution between organs etc., as in DAISY, WAVE and RZWQM or estimated indirectly using predefined curves for potential N-uptake. In some models (eg SOIL-N), the maximum uptake is specified by the user. This may be an advantage in research studies where this component can be assessed, but it hampers the use of the model for predictive purposes. The other models contain or may be combined with a growth module. The N-uptake may be calculated on the basis of transpiration fluxes, assuming uniform concentrations at the root surface and in the bulk soil (ANIMO), or by calculating transport of water and solute from the bulk soil to the root (DAISY, RZWQM, and WAVE)

The plant growth modules differ considerably among the model codes. ANIMO and LEACHM deal with plant processes only in a sketchy way, and SOIL-N hardly includes them. DAISY, WAVE and RZWQM simulate crop production while accounting for gross photosynthesis, respiration, distribution of dry matter and nitrogen between the different organs etc., though the types and number of available crop modules varies.

The general conclusion from the test of 14 model codes conducted by de Willigen

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(1991) was that prediction of nitrogen uptake by crops and dry-matter production requires one of the model codes containing a detailed growth module. However, for both soil water and mineralization simulations, the results showed that the detailed mechanistic model codes were not necessarily better than simple models. They require detailed information about soil hydraulic and chemical properties, and are very sensitive to parameter values. On the other hand they apply to a wider range of conditions than more simple models do.

2.3. PHOSPHOROUS MODELLING CODES

Traditionally, losses of phosphorous to groundwater and surface water from non-point sources have been regarded as a minor problem compared to the loads arising from point sources such as urban sewage discharge. However, due to large efforts put into controlling these point sources during recent years the relative load from non-point sources is increasing. Additionally, the magnitude of these loads has periodically been found to be rather large (Culley, 1983; Schjønning et al., 1995). Phosphorous has been identified as being the limiting nutrient for the primary production in many North European lakes, and during spring it may also be the limiting factor in coastal areas. The ability to control losses of phosphorous from non-point sources is therefore crucial for the water quality in these compartments.

Phosphorous is a key plant nutrient which is applied to agricultural areas either in mineral fertilizers or through organic manure. There are two main processes responsible for transport of phosphorous to surface waters and groundwater. The primary process is surface transport of particulate bound phosphorous driven by hydraulic soil erosion. The second process is vertical transport of soluble inorganic or organic and/or particulate phosphorous through preferential pathways in the upper soil to drains and groundwater. The significance of this latter pathway and the relative importance of transport of viz. solute and particulate phosphorous through the unsaturated soil column is generally not well known. However, as Phosphate is expected to be rather mobile under saturated and hereby reducing conditions, leaching of solute phosphorous out of the root zone may under certain conditions, eg local saturation in the upper soil layers, contribute significantly to the total losses.

Generally, the level of development of phosphorous models is lower than for nitrogen and pesticide models, and the main efforts have been put into modelling of the overland flow processes, which primarily involve transport of particulate bound phosphorous during heavy rainfall events causing erosion. Transport of soluble phosphorous is considered to occur to a less extent and is depending on the desorption processes. As the processes responsible for the overland transport of phosphorous are distributed in nature, point/field scale representation of these processes are very simplistic. An example of a field scale model is EPIC (Sharpley and Williams, 1990; Williams, 1995) which uses the SCS-curve method to estimate runoff of water and sediment on a standardised hill slope. An example of a distributed approach for modelling of the transport of phosphorous in surface runoff is the phosphorous module developed for the ANSWERS model code (Storm et al. 1988) which is an event based description of the transport of particulate bound and soluble phosphorous along with

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surface runoff of water. The solube fraction of phosphorous is calcultated as nonequilibrium desorption from the soil surface to the runoff water.

With respect to the vertical representation of the chemical processes involving phosphorous the existing model codes differ in complexity and in the assumptions made when simulating the complex nature of phosphorous cycling. Examples of model codes aiming to describe the chemical reactions of phosphorous in an unsaturated soil column are EPIC, ANIMO-P (Rijtema et al. 1991) and the approach by van der Zee and Gjaltema (1992).

2.4. PESTICIDE MODELLING CODES

During recent years pesticide losses from rural areas have been recognised as a major problem. Several different pesticides have been detected in groundwater and surface waters in many countries (Fielding et al. 1990), revealing a demand for reassessment of previous and present management practices. Especially, findings located in deep groundwater aquifers usually considered to be protected by impermeable clay layers (Brüsch and Kristiansen, 1994) have cast doubt on the knowledge and assumptions associated with the current procedure for registration and approval of pesticides.

The numerous findings have also revealed a need for development of predictive tools capable of quantifying the transport processes in order to perform risk assessment. This has converted the objective of model development from a research level to a functional level, and put more focus on reliability and validation.

In Europe, use of models in the pesticide registration has been included in the legislation through EEC-directive (91/414), regarding "uniform principles", stating that use of numerical models shall be incorporated into the registration procedure. At present only two countries, The Netherlands and Germany, have implemented model simulations in the standard procedure. The remaining countries are awaiting the results and recommendations from an EU working group regarding selection and use of pesticide fate models (FOCUS, 1995).

At the moment, many different models exist, claiming to be able to describe the fate of pesticides from application on the soil surface and through the unsaturated zone, hereby predicting the final load to the groundwater. Some models also consider surface runoff, and very few contain descriptions of lateral transport in the saturated zone.

In general, the descriptions of the transformation processes i.e. sorption and degradation differ in complexity and hereby also in parameter requirement. For instance, in some model codes the degradation rate is allowed to be specified differently with depth, phase (solid/liquid), site (matrix/macropores) or reaction type (eg hydrolysis, photolysis, biodeg.). However, the input parameters required for such complex decriptions are usually not available (Styczen and Villholth 1994, Bosch and Boesten, 1994a). This implies a high degree of uncertainty on the simulation results and makes calibration necessary.

Additionally, does the selection of appropriate input parameters and the following interpretation of the simulation results suffer from lack of knowledge regarding the variation in pesticide related parameters in the soil. Large variation in eg sorption coefficients and degradation rates for various pesticides has been observed at different sites, soil types and depths.

In general, testing and validation of models under a variety of conditions still remain to be performed. Some examples of model evaluations and comparisons are described in Pennel et al. 1990, Jarvis et al. 1994, Styczen and Villholth 1994, Bosch and Boesten, 1994b.

A review of selected approaches describing processes known to significantly influence the fate of pesticides in the unsaturated zone are shown in Table 3.

The main differences between the described modelling codes are their ability to mimic various hydrological conditions and their complexity with respect to chemical transformation processes. In order to take part in a standard registration procedure, it is important that the model codes are able to simulate different scenarios known to represent the variation in hydrology in the area under consideration. As an example which is valid for Danish conditions, this implies that the codes should provide various options for selection of the lower boundary conditions such as groundwater present in the root zone, and that special features like subsurface drainage can be included. One of the described model codes (PELMO) only contain the lysimeter boundary as an option while the number of available boundary conditions in the other model codes range from 4 to 9. The option for including subsurface drainage is only provided by MACRO, RZWQM and MIKE SHE.

The most common approach for simulating chemical degradation reactions is to assume one type of reaction described as 1. order decay and allowed to depend on temperature and soil moisture content. Some codes however (RZWQM and LEACHM), consider different degradation reactions, and MACRO allows different reaction rates to be associated with the matrix and the macropores even though the different degradation rates required as input are not commonly available, making parameter assessment difficult. The same problem is identified in the descriptions of the sorption processes. PESTLA, for instance, allows for kinetic sorption which require input of a sorption rate.

The general conclusion from the evaluation of existing pesticide modelling codes is that they provide strong and useful tools for research and comparative risk assessment, but that the present validation status is not adequate for predicting environmental concentrations (PEC) under different hydrological conditions for legislative purposes.

3. Modelling at Field and Catchment Scale

3.1. PROBLEMS AND APPROACHES IN UPSCALING

The modelling approaches presented in the previous sections focused on describing the transport processes in single soil columns, also regarded as point scale approaches.

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Table 3. Review of some of the most comprehensive pesticide modelling codes.

Process	PELMO	PESTLA	MACRO	LEACHM	RZWQM	WAVE	MIKE SHE
Water flow	Capacity model	Richards' eqn.	Richards' eqn. in matrix	Richards' eqn.	Green & Ampt/ Richards' eqn.	Richards' eqn. in matrix	Richards' eqn. in matrix
Pref. flow	-	-	+	-	+	-(+)	+
Lower boundary	Free drainage	Daily groundwater level, daily flux, predescribed flux, daily potential, zero flux, free drainage.	Free drainage, constant potential, constant hydraulic gradient, groundwater in root zone or zero flux	Constant potential, free drainage, zero flux, lysimeter	Free drainage, constant potential	Free drainage, gwt., lysimeter, fluxes, pressure head	Free drainage, constant potential, predescribed flux, groundwater in root zone, daily ground water level.
Solute transport	Piston displacement	Conv./disp. eqn.	Conv./disp. eqn.	Conv./disp. eqn.	Piston displacement, + mobile/immobile water	Conv./disp. eqn. + mobile/immobile water	Conv./disp. eqn.
Heat	Empirical f(T_{air}, depth)	Heat flux calc. f(T_{air}, heat conductivity, $\delta T/\delta z$)	Heat flux calc. f(T_{air}, heat conductivity, $\delta T/\delta z$)	Heat flux calc. f(T_{air}, θ, heat conductivity, $\delta T/\delta z$)	Heat flux calc. f(T_{air}, heat conductivity, $\delta T/\delta z$)	Heat flux calc. f(T_{air}, heat conductivity, $\delta T/\delta z$)	Empirical f(T_{air}, depth)
Sorption	Linear, Freundlich, kinetic, f(depth)	Linear + Freundlich	Linear, K_d defined in depth.	Linear, Freundlich, two site linear: equilibrium/kinetic	Two sites: viz. linear and kinetic	Two sites, linear + kinetic.	Linear, Freundlich, Langmuir. Kd defined in depth.
Degradation	n. order, f(θ, T, depth)	1. order, f(θ, T, depth)	1. order, f(θ, T), defined separately in depth, matrix/macropores, solid/liquid phase.	1. order, f(T, θ). Defined separately in liquid/solid phase.	1. order, f(T, θ). Different reaction types.	1. order, f(θ, T).	1. order, f(θ, T). Defined separately in depth.
Volatil.	+	-	-	+	+	+	-
Plant uptake	Passive, f(transpiration)	Passive, f(transpiration)	Passive, f(transpiration)	Passive, f(transpiration)	Passive, f(transpiration)	Passive, f(transpiration)	Passive, f(transpiration)
Other functions included			Subsurface drainage. Swell/shrink of macropores. Switch between 1 & 2 domains.	Simultaneous simulation of several compounds.	Includes biodegradation, oxidation, complexation, photolysis, hydrolysis, bicarbonate buffering.	Hysteresis	Subsurface drainage. Three-dimensional ground water and surface runoff can be simulated at catchment scale.
References	Klein (1993)	Boesten and van der Linden (1991), Boesten (1993)	Jarvis (1991, 1994)	Wagenet and Hutson (1987), Hutson and Wagenet (1992)	DeCoursey et al. (1989, 1992)	Dust et al. (1994), Vereecken et al. (1994), Vanclooster et al. (1994)	Abbott et al. (1986), Refsgaard and Storm (1995)

(+): Planned in future versions, T: Temperature, θ: Soil moisture content, z: Depth,

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These models are valuable research tools for studying transport and transformation processes, but contain a range of limitations when it comes to predicting loads of agrochemicals to streams and aquifers arising from different agricultural systems. The models do not consider spreading processes arising from flux of water and solute in two or three directions in the groundwater zone, nor do they allow for considerations regarding the horizontal variation in hydraulic and chemical properties in the soil or the distributed nature of geology, topography, drainage networks and agricultural management.

For projects focusing on larger geographical areas, such as studies of the impacts of agricultural management practices on solute concentration in groundwater aquifers or streams, extrapolation based on small-scale studies becomes difficult because such studies are likely to represent only a limited selection of the characteristics (soil types, depth of unsaturated zone, vegetation, etc.) found in a larger area. Models covering larger areas may therefore provide a better basis for decision making with regard to management strategies or policies. On the other hand, the problem here is how adequately the study area should be characterized. As the scale increases, the information required for running the models cannot be derived directly (e.g. from measurements), and the results become only approximate due to the simplifications introduced, and the neglected spatial variability of certain features. In addition, model validation is difficult. Site-specific comparisons against observed data cannot be made because representative (or effective) parameter values rather than measured values are used. It is therefore of major importance that model users recognize and report the limitations and uncertainties in the model predictions.

As described in Chapter 4 of this book, numerical groundwater models describing the flow and transport mechanisms of aquifers have been developed since the 1970's and applied in numerous pollution studies. They have mainly described the advection and dispersion of conservative solutes. More recently, geochemical and biochemical reactions have been included to simulate the fate of reactive pollutants from point sources such as industrial and municipal waste-disposal sites (see Chapter 5 of this book). Few attempts have been made to simulate non-point pollution from fertilizers and chemicals used in agriculture. The main problem arises from the need for characterization of physical, chemical and biochemical properties of large areas. An additional problem in connection with groundwater modelling is to provide an estimation of the solute input from the unsaturated zone to the groundwater. If the estimates are not based on results from leaching models, the timing and volume of nitrate fluxes are difficult to assess, because they depend on several factors, such as the depth of the unsaturated zone. This will have an important effect on the simulated concentrations in the groundwater. In areas with a shallow groundwater table, the surface application is reflected in the temporal variation of nitrate concentrations in the groundwater to a higher degree than in areas with large distances to the groundwater table.

Two principally different approaches for upscaling simulations of agricultural managements systems have been developed during the recent years. As these two approaches have similarities with two of the classes of hydrological models described in Chapter 2 of this book they will be denoted lumped conceptual and distributed physically-based approaches, respectively.

In the lumped conceptual approach the area under consideration, typically an agricultural field or a small catchment, is conceptualized as having horizontally homogenous soil properties, uniform rainfall distribution and only one type of landuse and management practice each year. The process descriptions related to the transport and transformation processes in the root zone correspond to those applied in the single column models, whereas the components describing surface run off usually are extended in order to handle overland flow and erosion as functions of areal and topographical data. Percolation from the root zone is routed to the groundwater, but lateral groundwater flow is typically not considered. Similarly, feedback from groundwater zone to unsaturated zone is usually not included.

The distributed physically-based approach allows for horizontal distributions of physical and chemical parameters, rainfall, land use, topography etc. Lateral surface and subsurface flows are included.

In the following two subsections examples of the two model types are briefly introduced and intercompared.

3.2. LUMPED CONCEPTUAL FIELD SCALE MODELLING

Use of the lumped approach for modelling agricultural aspects on larger scales involves conceptual representation of the area in question as being homogenous with respect to climatic conditions, topography, geology, land use, management practice, soil characteristics etc.. This implies that the hydrological unit in this approach is representing a field, hillslope or subcatchment.

Most existing model codes of this type use a simple hydrological model for description of the water balance such as the capacity approach for vertical water flow. The major reason for this is that the spatial resolution and assumptions of the lumped approach are too coarse basis for more complex physically based modelling. The simple water balance approach does not allow direct interaction between groundwater and surface waters. Only fluxes out of the conceptual hydrological unit are accounted for and are routed to rivers or groundwater based on empirical equations. Fluxes between single units and local phenomena such as periodically ponding due to inhomogeneous topography or soil characteristics are not accounted for.

Examples of lumped conceptual models used for assessment of agricultural systems are the family of model codes developed by the U.S. Department of Agriculture having the hydrology in terms of the modified SCS curve number technique in common. These models are CREAMS (Knisel et al., 1980; Knisel and Williams, 1995) which contain fairly simple descriptions of water, nitrate and pesticide transport, GLEAMS (Leonard et al. 1987) which has more chemical focus on pesticides and latest SWRRB (Arnold et al. 1990; Arnold and Williams, 1995), which is described as a basin scale model for simulation of water, nitrate, phosphorous and pesticides. SWRRB include the hydrology part of CREAMS and the pesticide part of GLEAMS and allow for simultaneous calculations of hydrological units representing different fields or subcatchments differing in eg landuse, and agricultural management practice.

Some of the models operate on a continuous simulation basis, while others, such as

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ANSWERS (Beasly et al. 1980) and AGNPS (Young et al. 1995) can only simulate single events. For the event models the very difficult estimation of catchments initial conditions are crucial for obtaining reliable model predictions.

3.3. DISTRIBUTED PHYSICALLY-BASED CATCHMENT SCALE MODELLING

Using a distributed physically-based model for agricultural impact assessment provides the possibility of including the distributed nature of different agricultural management practices within an entire catchment, and allow for detailed descriptions of water and solute fluxes within the catchment. The hydrological unit in this approach consists of a large number of internal grids which are defined independently of the catchment structure with a spatial resolution reflecting the complexity of the catchment area in terms of spatial distribution in climatical conditions, topography, geology, land use, agricultural management practice etc,. The choice of grid size is defined by the modeller allowing for simulations varying in detail and complexity, depending on availability of model parameters and the objective of the study. Fluxes of water and solutes are routed between the internal grid elements depending on the spatial representation of catchment, and the physically based nature of the approach also allow for simulation of the interaction with groundwater and surface water.

This type of modelling approach require a large number of parameters describing the spatial variation within an entire catchment. If such parameters are not available application of a lumped conceptual approach may be adequate. However, especially for studies involving detailed assessment of solute fluxes and concentrations, the spatial representation and the physically based nature of the calculations is crucial.

An example of a fully distributed model is the MIKE SHE (Refsgaard and Storm, 1995)

Examples of studies attempting to couple the unsaturated and saturated zone models in order to perform assessments of the impact of agricultural management on the nitrate load to streams and aquifers, are described in Bogardi et al. (1988), Storm et al. (1990), Styczen and Storm (1993).

4. Case study: Modelling of Nitrogen Transport and Transformation on a Catchment Scale

4.1. INTRODUCTION

The present case study is one of the outputs from a comprehensive Danish research and development programme (1986-90), which was carried out with the aim of studying the pollution from nutrients and organic matters in agriculture. The research programme was multidisciplinary and involved a large number of research institutions. It included field investigations, process studies and modelling.

The present case study briefly describes a distributed hydrological modelling of nitrate transport and transformation for the 440 km^2 Karup River catchment. The nitrogen modelling covers the entire land phase of the hydrological cycle - from the

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source on the soil surface, through the soil zone and the groundwater to the streams. The modelling was based on the MIKE SHE for catchment processes and for the DAISY model (Hansen et al. 1991) for simulation of the nitrogen dynamics in the root zone. The concepts in the coupling of the one-dimensional leaching model (DAISY) and the three-dimensional model (MIKE SHE) is illustrated in Fig. 1. A more detailed description of the present case study is presented by Styczen and Storm (1993).

4.2. MODEL SET-UP

The Karup catchment was represented in a three-dimensional network. The discretization is 500 m in the horizontal directions and varies in the vertical from 5 to 40 cm in the unsaturated zone, and 5 m in the permanently saturated zone. Information on soil and vegetation properties were collected and processed based on information from a number of wells, a three-dimensional geological map was superimposed on the model grid to provide the hydrogeological parameter values. The topography and the river network have been digitized, and all relevant climatological data collected. The overall land use has been identified.

4.3. RESULTS

4.3.1 Discharge and groundwater table hydrographs

The streamflow is simulated for the period 1969 - 1988 at several sites. A comparison with measured discharge at the catchment outlet is shown for four years in Fig. 2. In addition the simulated groundwater table is compared with observations in selected wells (Fig. 3). The comparison indicates that the modelling system simulates the hydrological regime with acceptable accuracy.

4.3.2 Leakage from the root zone

To simulate the trend in the nitrate concentrations in the groundwater and the streams, it is necessary to have information on the history of the fertilizer application in space and time. This information is difficult to obtain in details. for example it is not possible to estimate which type of crop was growing on one particular field in one particular year in the past. The most detailed information one can expect to obtain is a spatial percentage of the various crops, and the types of farming practices that have been carried out in the area. Based on this information a series of 14 crop rotation schemes covering the period of interest was established, and at random distributed over the area.

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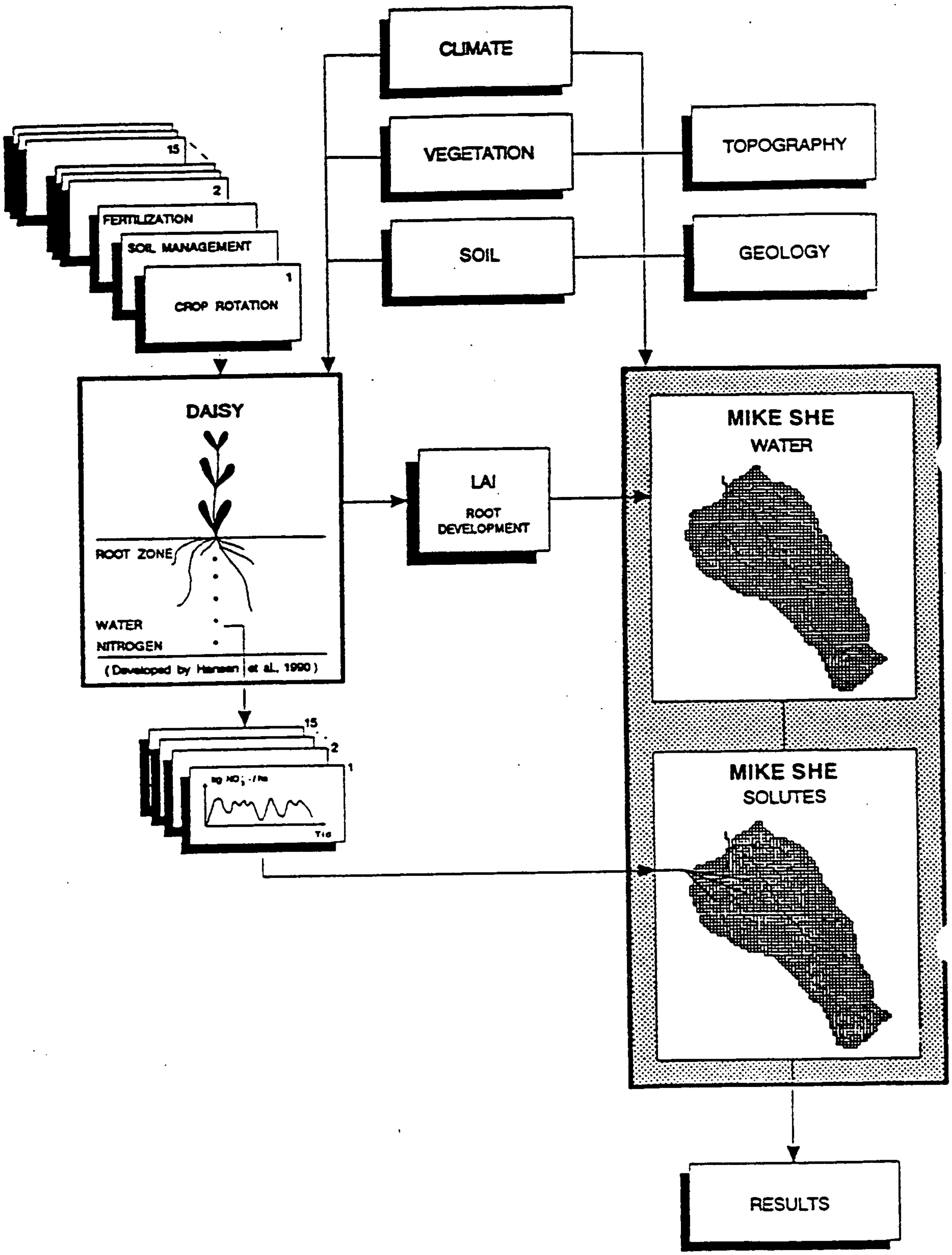

Figure 1. Coupling of the one-dimensional leaching model (DAISY) and the three-dimensional model (MIKE SHE)

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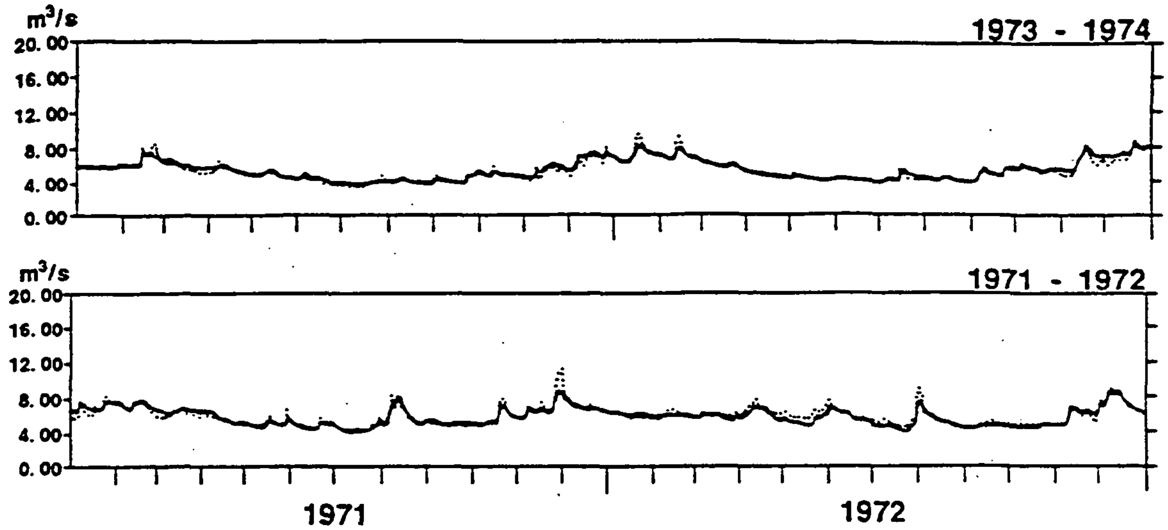

Figure 2. Comparison between simulated and observed river runoff for the period 1971-74

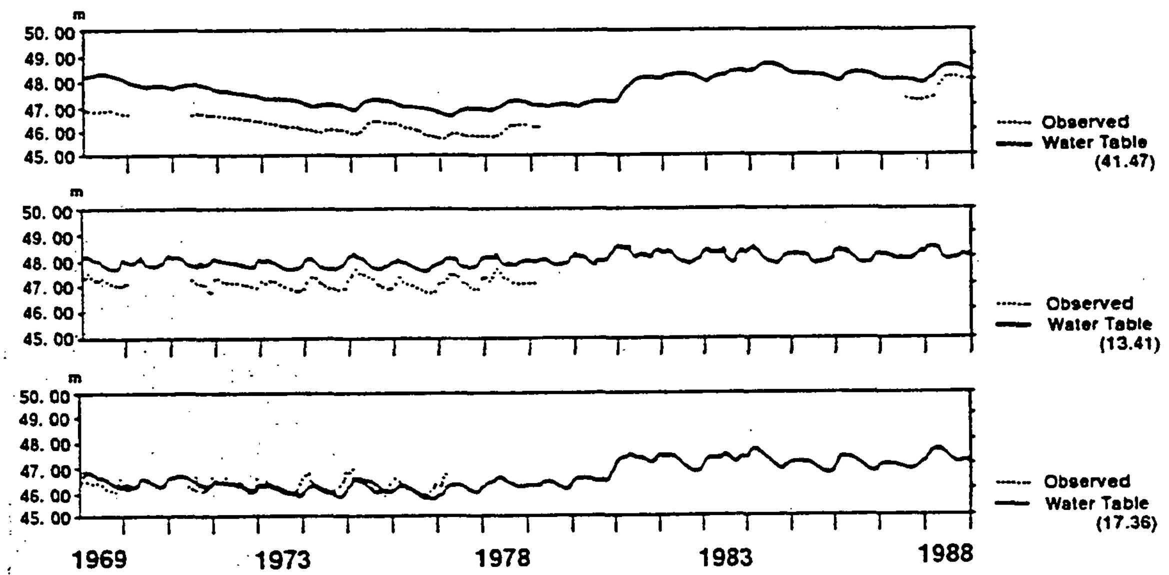

Figure 3. Comparison between simulated and observed groundwater table time series in selected wells

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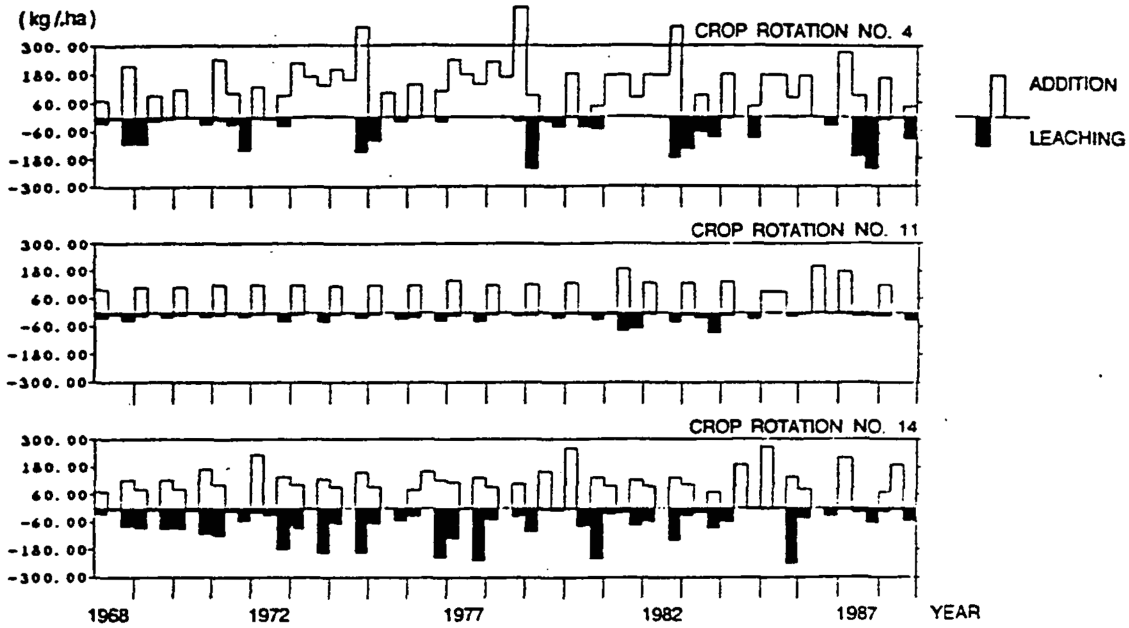

Figure 4. Nitrate leaching (NO_3^--N) from three of the crop rotations calculated by DAISY and summarized over four-months periods. The shown additions of N only include mineral fertilizer and the already mineralized part of manure.

Based on estimated application rates of organic and mineral fertilizer to the individual crops each year, the DAISY model simulates the crop growth, root uptake, mineralization and leakage of nitrate from the root zone. Fig. 4 shows time series of application and leakage for selected crop rotation schemes. On farms which are based on mainly meat production a large amount of organic fertilizer will often be applied on the fields in the autumn. In this period there is a potential risk for significant losses to the groundwater system.

4.3.3 Nitrate concentrations in groundwater

While the root zone model simulates one 'soil column' at a time the total model allows studies of the variations in space and time at regional scale. Fig. 5 illustrates the variation in simulated NO_3^--concentrations in the upper groundwater layer of the Karup catchment below three selected cropping schemes for two points with different depths of the unsaturated zone. A deep unsaturated zone is seen to dampen the influence of a single year.

Fig. 6 shows the spatial variation in simulated NO_3^--concentrations in the upper groundwater layer at a specific time. The very large variation of concentration both in space and time is noticed.

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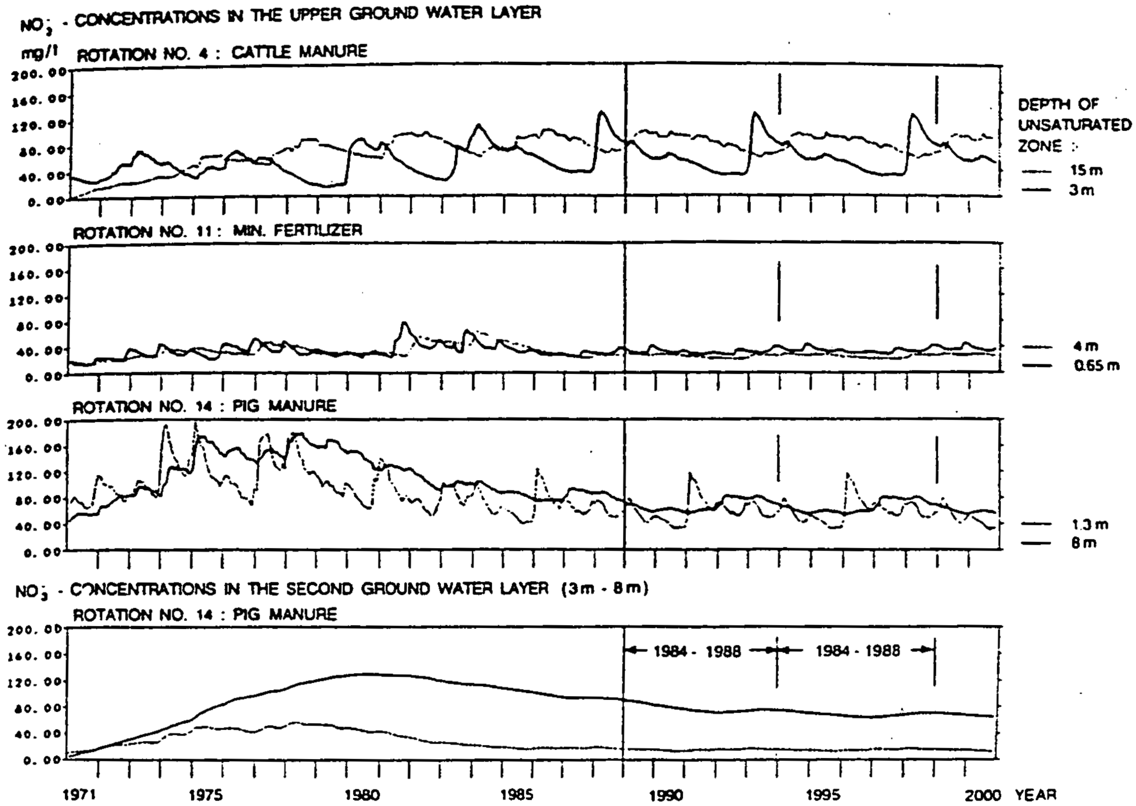

Figure 5. Temporal variation in NO_3^- concentrations in the upper groundwater layer beneath three selected rotation schemes, with two different distances to the groundwater table. The data are extracted from selected grids (not averaged).

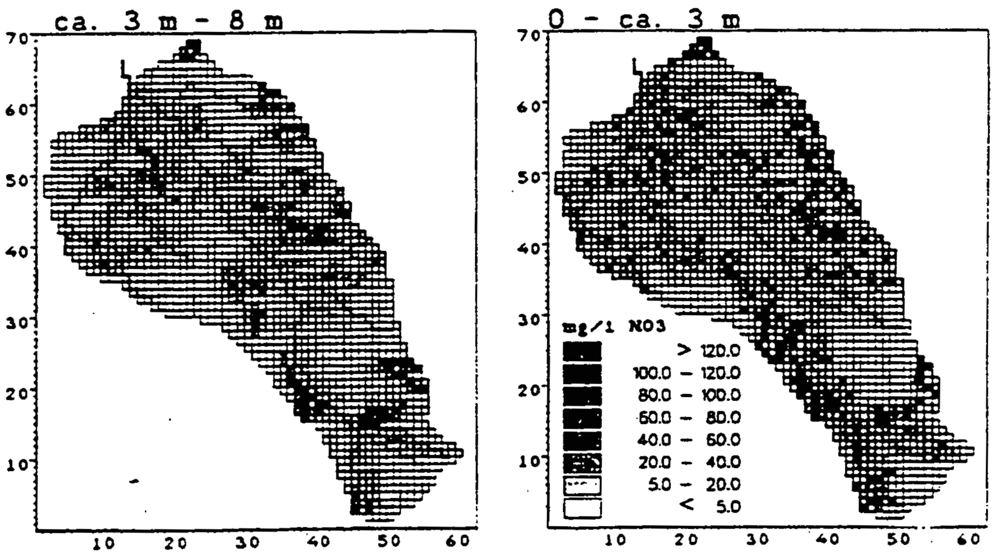

Figure 6. Spatial variation in NO_3^- concentrations in the upper groundwater layer over the entire catchment at a specific time.

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5. Discussion on Model Applicability and Limitations

Compared to 'pure' hydrological modelling the agrochemical modelling has naturally not reached the same level of reliability and applicability. The issue is much more complex because it, in addition to all the inherent hydrological problems, comprises agrochemical problems with regard to process understanding, field data availability and modelling methodology. On the basis of the significant progresses made during the past decade and the increasing demand for agrochemical modelling, very significant progress can, however, be expected during the coming years.

The key problems which need to be addressed are insufficient understanding of processes at a local scale, insufficient availability of field data and inadequate methodology for treating the effects of heterogeneity of process parameters and variables.

Many of the process descriptions, even in the so called 'physically-based' model codes, comprise a theoretically based frame, but include empirical equations, which cannot be parameterized further until more knowledge on processes or more particular field data become available. This is similar to the situation on soil erosion modelling, where the importance and difficulties in describing local scale processes are outlined in Chapter 6 of this book.

The most significant progress made in agrochemical modelling during the past decade is related to development of comprehensive physically-based leaching models. In spite of considerable uncertainty involved due to the deficiencies described above, such models appear to have some predictive capability, and can, with professional and cautious use, be very useful tools for management purposes.

For catchment scale modelling the present status is less advanced. A key problem in this respect is the very large spatial (and temporal) variability within a catchment of important parameters, such as soil hydraulic, soil chemical, geological and topographical parameters as well as cropping pattern, fertilization practise, tillage etc. The existing lumped conceptual models are rather easily operational; but have significant theoretical limitations and are, due to their limitations not able to address the effects of many key management options. The distributed physically-based models, based on the advanced (point scale) leaching models in a distributed hydrological modelling framework have significant potentials. The few existing examples of such approaches, such as the one described in Section 4 above, have indicated the usefulness of the approach, but have not highlighted the inherent difficulties. The key fundamental problem in this regard relates to the problem of using the point scale leaching models at grid scales in the distributed models. In the above example for the 440 km^2 catchment the grid size was 25 ha. Such discretization is sufficient for producing a good hydrological simulation of discharges and groundwater levels, but may be problematic for simulation of nitrogen and pesticide leaching, transport and transformation. In connection with many policy questions information at national scale or even above (e.g. EU) is relevant. In such cases the discretization may, in practise, have to be even larger. Fundamental research on these issues is required in the coming years.

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CHAPTER 12
AN ENGINEERING CASE STUDY - MODELLING THE INFLUENCES OF GABCIKOVO HYDROPOWER PLANT ON THE HYDROLOGY AND ECOLOGY IN THE SLOVAKIAN PART OF THE RIVER BRANCH SYSTEM OF ZITNY OSTROV

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1. Introduction

1.1. THE DANUBIAN LOWLAND AND THE GABCIKOVO HYDROPOWER SCHEME

The Danubian Lowland (Fig. 1) between Bratislava and Komárno is an inland delta formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which throughout the year receives in the order of 30 m^3/s infiltration water from the Danube in the upper parts of the area and returns it into the Danube and the drainage channels in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

Human influence has gradually changed the hydrological regime in the area. Construction of dams upstream of Bratislava together with exploitation of river sediments has significantly deepened the river bed and lowered the water level in the river. These changes have had a significant influence on the conditions of the ground water regime as well as the sensitive riverside forests downstream of Bratislava. In spite of this basically negative trend the floodplain area with its alluvial forests and the associated ecosystems still represents a very unique landscape of outstanding importance.

The Gabcikovo hydropower scheme was put into operation in 1992. A large number of hydraulic structures has been established as part of the hydropower scheme. The key elements are a system of weirs across the Danube at Cunovo 15 km downstream of Bratislava, a reservoir created by the damming at Cunovo, a new lined canal running parallel to the Old Danube over a stretch of approximately 30 km for navigation and with intake to the hydropower plant, a hydropower plant and two shiplocks at Gabcikovo, and an intake structure at Dobrohost diverting water from the new canal to the river branch system. The entire scheme has significantly affected the hydrological regime and the ecosystem of the region. The scheme was originally planned and the major parts of the construction were carried out as a joint effort between Czecho-

M. B. Abbott and J. C. Refsgaard (eds.), Distributed Hydrological Modelling, 233–253.

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Slovakia and Hungary. However, today Gabcikovo is a matter of controversy between Slovakia and Hungary, who have referred some disputed questions to the International Court of Justice in Haag.

Comprehensive monitoring and assessments of environmental impacts have been made, see Mucha (1995) for an overview.

1.2. THE PHARE PROJECT 'DANUBIAN LOWLAND - GROUND WATER MODEL'

To understand and analyze the complex relationships between physical. chemical and biological changes in the surface- and subsurface water regimes in the Danubian Lowland requires multidisciplinary expertise in combination with field data and advanced mathematical modelling techniques. For this purpose the project "Danubian Lowland - Ground Water Model" was defined within the PHARE programme agreed upon between the European Commission and the Government of the Slovak Republic.

The overall project objective was to establish a comprehensive modelling and information system suitable as a decision support tool for water resources management in the area. The aim of the developed integrated modelling system was to provide a reliable tool for analyzing the environmental impact of alternative management strategies and hence support in the formulation of optimal management strategies leading to a protection of the water resources and to a sound ecological development for the area.

The PHARE project was executed by the Slovak Ministry of the Environment. Specialists from the following Slovakian organisations were involved in various aspects of the project implementation: Comenius University, Faculty of Natural Science (PRIF UK); Water Research Institute (VUVH); Irrigation Research Institute (VUZH); and Ground Water Consulting, Ltd (GWC).

A Danish-Dutch consortium of six organisations, headed by DHI, was selected as consultant for the project. The project was initiated in the beginning of 1992 and was completed by the end of 1995.

1.3. THE PRESENT CHAPTER

The aims of the present chapter are to illustrate the overall structure and functioning of the developed integrated modelling and information system, to outline the modelling approach for such complex case and to show some selected applications from the flood plain and river branch system.

A key for analyzing the impacts on the water resources of the changed surface water conditions caused by the Gabcikovo plant is an integrated description of the river-aquifer system. The chapter provides details on the modelling of the river branch system and the aquifer system, which is being carried out using a newly developed full coupling between DHI's two modelling systems for rivers and hydrology, MIKE 11 and MIKE SHE, respectively.

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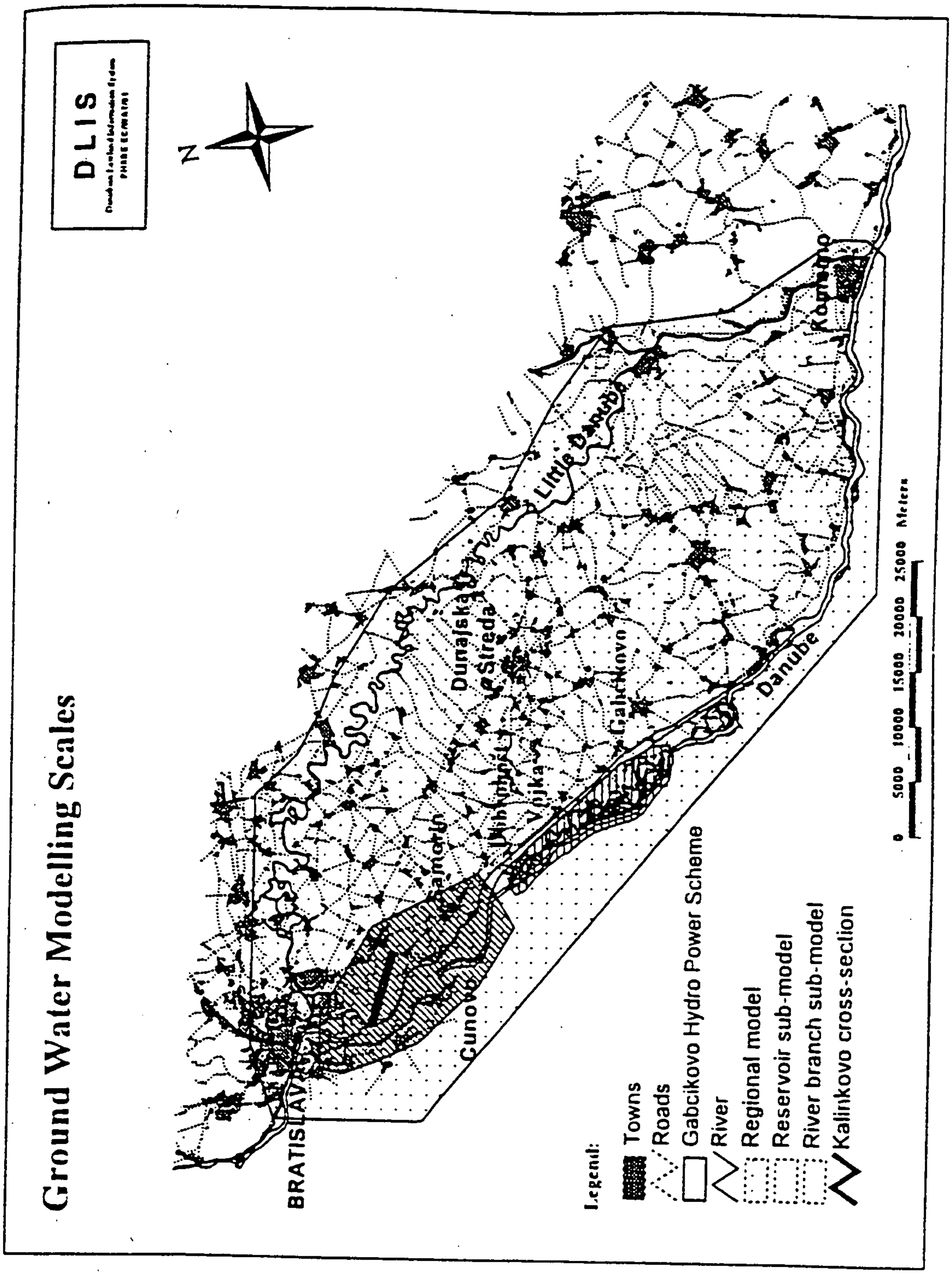

Figure 1. The Danubian Lowland area, layout of the Gabcikovo scheme and the extent of the regional model as well as of local modelling scales.

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2. Modelling approach

2.1. INTEGRATED MODELLING SYSTEM

In order to address the problems within the project area an integrated modelling system (Fig. 2) has been established by combining the following existing and well proven mathematical modelling systems:

* *MIKE SHE* (Refsgaard and Storm, 1995) which, on catchment scale, can simulate the major flow and transport processes of the hydrological cycle which are traditionally divided in separate components:
  - 1-D flow and transport in the unsaturated zone
  - 3-D flow and transport in the ground water zone
  - 2-D flow and transport on the ground surface
  - 1-D flow and transport in the river.

  All the above processes are fully coupled allowing for feedbacks and interactions between components. In addition to the above mentioned components, MIKE SHE includes modules for multi-component geochemical and biodegradation reactions in the saturated zone (Engesgaard, Chapter 5).
* *MIKE 11* (Havnø et al., 1995), which is a one-dimensional river modelling system. MIKE 11 is used for hydraulics, sediment transport and morphology, and water quality. MIKE 11 is based on the complete dynamic wave formulation of the Saint Venant partial differential equations. The modules for sediment transport and morphology are able to deal with cohesive and non-cohesive sediment transport, as well as the accompanying morphological changes of the river bed. The non-cohesive model operates on a number of different grain sizes, taking into account shielding effects.
* *MIKE 21* (DHI, 1995), which is a two-dimensional hydrodynamic modelling system. MIKE 21 is used for reservoir modelling, including hydrodynamics, sediment transport and water quality. The sediment transport modules deals with both cohesive and non-cohesive sediment, and the non-cohesive module operates on a number of different grain size fractions.
* MIKE 11 and MIKE 21 include *River/Reservoir Water Quality (WQ) and Eutrophication (EU)* (Havnø et al., 1995; VKI, 1995) modules to describe oxygen, ammonium, nitrate and phosphorus concentrations and oxygen demands as well as eutrophication issues such as bio-mass production and degradation.
* *DAISY* (Hansen et al., 1991) is a one-dimensional root zone model for simulation of soil water dynamics, crop growth and nitrogen dynamics for various agricultural management practices and strategies. The particular processes considered include transformation and transport involving water, heat, carbon and nitrogen.

The integrated modelling system is formed by the exchange of data and the feed-backs between the individual modelling systems. The structure of the integrated modelling system and the exchange of data between the various modelling systems are illustrated in Fig. 2. The interfaces A-E between the various models are briefly described below:

A) MIKE SHE forms the core of the integrated modelling system having interfaces

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to all the individual modelling systems. The coupling of MIKE SHE and MIKE 11 is a fully dynamic coupling where data is exchanged after each computational time step. This interface is described in more details below.

The remaining modelling systems are coupled in a more simple manner involving a sequential execution of individual models and subsequently a transfer of boundary conditions from one model to another. Some examples are listed below.

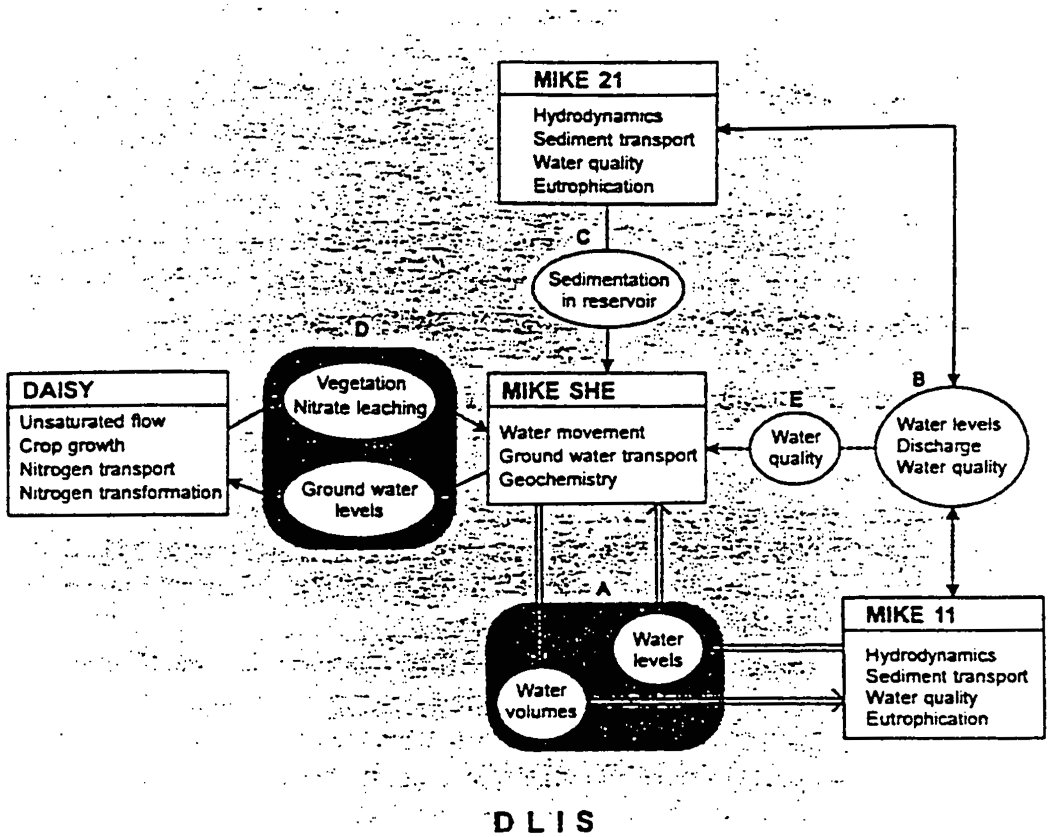

Figure 2. Structure of the integrated modelling system with indication of the interactions between the different models.

B) Results of eutrophication simulations with MIKE 21 in the reservoir are used to estimate the concentration of various water quality parameters in the water that enters the Danube downstream of the reservoir to be used for water quality simulations for the Danube using MIKE 11.

C) Sediment transport simulations in the reservoir with MIKE 21 provide information on the amount of fine sediment on the bottom of the reservoir. This information is used to calculate leakage coefficients which are used in ground water modelling with MIKE SHE to calculate the exchange of water between the reservoir and the aquifer.

D) The DAISY model calculates vegetation parameters which are used in MIKE SHE

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to calculate the actual evapotranspiration. Ground water levels calculated with MIKE SHE act as lower boundary conditions for DAISY unsaturated zone simulations. Consequently, this process is iterative and requires a few model simulations.

E) Results from water quality simulations with MIKE 11 and MIKE 21 are used to estimate the concentration of various species in the water that infiltrates to the aquifer from the Danube and the reservoir. This is being used in the ground water quality simulations (geochemistry) with MIKE SHE.

The Danubian Lowland Information System (DLIS) is a combined data base and geographical information system which has been developed under this project. The DLIS is based on Informix (database) and Arc/Info (GIS) and provides a framework for data storage, maintenance, processing and presentation. In addition, an interface between DLIS and MIKE SHE allowing import and export of maps and time series files in MIKE SHE file formats has been established.

With regard to simulation of floodplain hydrology and ecology the core of the integrated modelling system is constituted by the MIKE SHE, the MIKE 11 and the newly developed, full coupling of the two systems described below.

2.2. A COUPLING OF MIKE SHE AND MIKE 11

The focus in MIKE SHE lies on catchment processes with a comparatively less advanced description of river processes. In contrary MIKE 11 has a more advanced description of river processes and a simpler catchment description than MIKE SHE. Hence, for cases where full emphasis is needed for both river and catchment processes a coupling of the two modelling systems is required.

A full coupling between MIKE SHE and MIKE 11 has been developed (Fig. 3). In the combined modelling system, the simulation takes place simultaneously in MIKE 11 and MIKE SHE, and data transfer between the two models takes place through shared memory. MIKE 11 calculates water levels in rivers and floodplains. The calculated water levels are transferred to MIKE SHE, where flood depth and areal extent are mapped by comparing the calculated water levels with surface topographic information stored in MIKE SHE. Subsequently, MIKE SHE calculates water fluxes in the remaining part of the hydrological cycle. Exchange of water between MIKE 11 and MIKE SHE may occur due to evaporation from surface water, infiltration, overland flow or river-aquifer exchange. Finally, water fluxes calculated with MIKE SHE are exchanged with MIKE 11 via source/sink terms in the continuity part of the Saint Venant equations in MIKE 11.

The MIKE SHE-MIKE 11 coupling is crucial for a correct description of the dynamics of the river-aquifer interaction. Firstly, the river width is larger than one MIKE SHE grid, in which case the MIKE SHE river-aquifer description is no longer valid. Secondly, the river/reservoir system comprises a large number of hydraulic structures, the operation of which cannot be accounted for in MIKE SHE. Thirdly, the very complex branch system with loops and flood cells needs a very efficient hydrodynamic formulation such as MIKE 11's.

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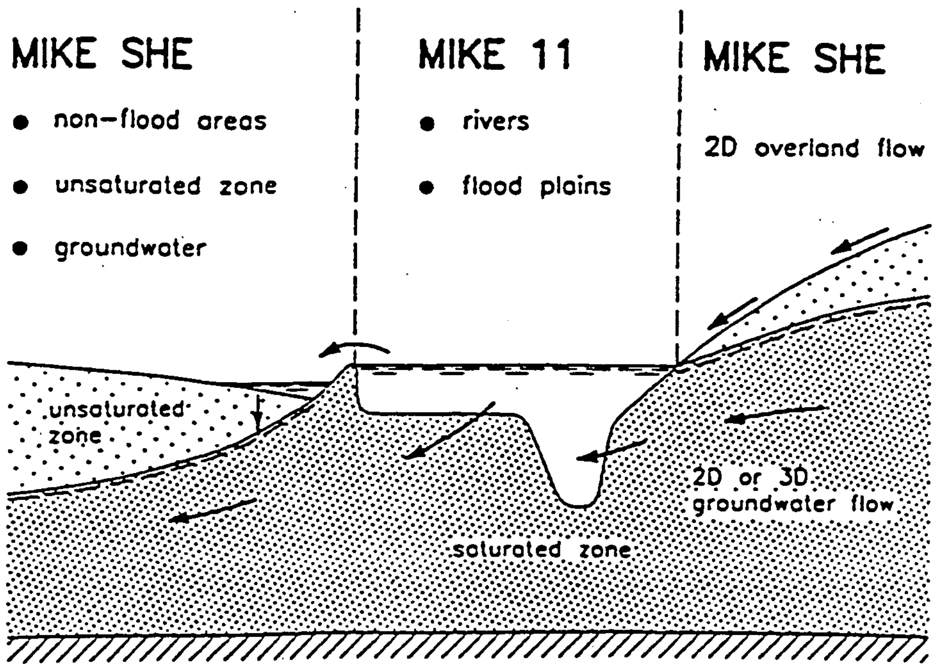

Figure 3. Structure of the MIKE 11 - MIKE SHE coupling.

2.3. MODELLING SCALES

As indicated in Fig. 1 modelling has been carried out at different spatial scales with different objectives. Thus, the following ground water models have been established:

- a regional ground water model for pre-dam conditions,
- a regional ground water model for post-dam conditions,
- a local ground water model for an area surrounding the reservoir (post-dam conditions),
- a local ground water model for the river branch system (both pre-and post-dam conditions), and
- a cross-sectional (vertical profile) model near Kalinkovo.

The regional model area covers about 3000 km^2. It applies a horizontal discretization of 500 m and in the vertical the aquifer has been divided into four geologically determined computational layers. The main objectives of the regional ground water modelling are to study the impacts of the damming of the Danube on the hydrological regime within the project area, in particular in terms of ground water levels and dynamics, and to provide reliable boundary conditions for local ground water models.

The ground water model for the reservoir area is a sub-model of the regional model and the parameter values of the local model are identical to the ones in the regional

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model with the only change that the local model applies a horizontal discretization of 250 m, and a finer vertical discretization with 7 computational layers. The objectives of the local ground water model around the reservoir are to provide more accurate results for this area than can be done with the more coarse regional model and to provide boundary conditions for the cross-sectional model.

Similarly, the local model for the river branch area is a sub-model of the regional model. This local model has a horizontal discretization of 100m. The objective of the local model for the branch system area is to make detailed predictions of the hydrological regime for alternative water management schemes and thus enable assessments of possible ecological changes in the floodplain area.

A 2 km long cross-sectional model near Kalinkovo with a horizontal discretization of 10 m and 24 vertical layers was established within the area of the local reservoir model in order to provide the hydraulic basis for comprehensive geochemical modelling.

2.4. PROCEDURES FOR MODEL CONSTRUCTION, CALIBRATION, VALIDATION AND APPLICATION

2.4.1 Model construction

All the applied models are based on distributed physically-based model codes. This implies that most of the required data for establishing the model setup and input data can be measured directly in the nature.

A setup of a MIKE 11 hydrodynamic model requires that the river geometry is known involving river cross-sections and various hydraulic structures in the system. A setup of the Hrusov reservoir in MIKE 21 requires that topographical information on the reservoir bottom (bathymetry) is available.

For the MIKE SHE hydrological modelling the same type of data are required, and in addition information on soils and geology is required as well as climatological and vegetation data.

Therefore, a setup for a physically-based model, as a minimum, always involves a geometrical description of the problem and some physical characteristics of the system. This could for instance be hydraulic conductivities in the saturated zone or roughness coefficients in river cross-sections and on the reservoir bottom.

In addition, the value of some state variables has to be known on the model boundaries (boundary condition). For hydrodynamic models the boundary conditions are always a combination of prescribed water levels and discharges. Ground water modelling involves, in principle, the same but described as ground water table or ground water flow. For water quality and geochemical modelling concentration of certain species, for instance nitrate and oxygen, must be known in the water that enters the system.

2.4.2 Model calibration

The calibration of a physically-based model implies that a sequence of simulation runs are carried out and model results are compared with measured data for a certain period

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of time.

MIKE 11 was calibrated against measurements of water levels and discharges. MIKE 21 was calibrated against flow velocities and MIKE SHE was calibrated against measured ground water levels, river flows and water levels.

When using physically-based models, the 'amount' of required calibration are reduced the more precise the geometry, the physical characteristics and the boundary conditions of the physical problem are described. In principle, if the model setup exactly reflects the real conditions no model calibration would be required at all. Obviously, this is a hypothetical situation and in practice some model calibration is always needed.

The available amount and quality of data within the project area provided a very good basis for model constructions. Almost all physical characteristics used in the model setup are based on measured data. Hence, the calibration of the various models has been performed by adjusting only a limited number of physical characteristics within a relatively narrow range.

For the calibration of the MIKE 11 hydrodynamic model the roughness of the river bed (the Manning number) was the main calibration parameter. For some structures the precise crest elevations was not known and the exact capacity of culverts was uncertain. Therefore, such geometrical data has been subject to limited calibration, but in general the geometry of the system was not adjusted during the calibration process.

For the MIKE 21 reservoir hydrodynamic model the main calibration parameters were the bottom roughness (Chezy number) and the eddy viscosity.

For the MIKE SHE ground water modelling the main calibration parameter was the hydraulic conductivity in the saturated zone. All other physical characteristics was, in general, kept at the measured values or at experience values from previous studies.

The DAISY unsaturated flow simulations were based on measured soil water retention curves and hydraulic conductivities.

2.4.3 Model validation

Because the calibration process involves some manipulation of parameter values good model results during a calibration process cannot automatically ensure that the model can perform equally well also for other periods. Therefore, model validations on independent data are required.

All the models have been validated by demonstrating the ability to reproduce measured data for a period outside the calibration period. For the MIKE SHE regional ground water flow models the model was even calibrated on pre-dam conditions and validated on post-dam conditions where the flow regime at some locations were significantly altered due to the construction of the reservoir and related hydraulic structures and canals.

2.4.4 Model application - integrated scenario simulations

The validated models were applied in a scenario approach simulating the hydrological conditions resulting from alternative possible operations of the entire system of hydraulic structures (alternative water management regimes). Thus, one historical (pre-

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dam) regime and three hypothetical water regimes corresponding to alternative operation schemes for the structures of the Gabcikovo system were simulated. Due to the integration of the overall modelling system (Fig. 2) each scenario simulation involves a sequence of model calculations. Interpretation of results are made for each single step describing the hydraulic/ecological/chemical conditions within a certain field of this study. However, the integrated modelling system is formed by the exchange of data and results between the various models. A typical integrated scenario simulation could involve the following model simulations:

Step 1) MIKE 11/MIKE 21 Hydrodynamic
Hydrodynamic simulations are carried out for the reservoir (MIKE 21) and for the rivers (MIKE 11) given a certain water management regime. Hydrodynamic modelling of the MIKE11 post-dam model provides boundary conditions for the reservoir MIKE 21 model.

Main output: Flow velocities and water levels in the reservoir, in the Danube and in the river branches.

Step 2) MIKE 21/MIKE 11 Water quality/Eutrophication/Sediment transport
Based on the simulated flow fields (step 1) sediment transport, water quality and eutrophication simulations are carried out. Eutrophication modelling of the reservoir provides concentration boundaries for water quality models for the downstream Old Danube and for eutrophication modelling of the downstream river branch system.

Main output: Amount of different sediment grain size fractions on the reservoir bottom and concentrations of oxygen and nitrate distributed in time and space.

Step 3) MIKE SHE/MIKE 11 regional ground water flow simulation
A ground water flow simulation for the entire model area is carried out using the coupled version of MIKE SHE and MIKE 11. The reservoir is included in the MIKE 11 hydrodynamic simulations in a simplified one-dimensional flow description. Based on information on the distribution of sediment on the reservoir bottom (step 2) leakage coefficients are calculated and subsequently used directly in the MIKE SHE ground water flow simulations. Here, the applied leakage coefficients play an important role for the exchange of water between the reservoir and the aquifer.

Main output: Ground water flow and ground water levels distributed in time and space.

Step 4) MIKE SHE/MIKE 11 local reservoir model simulation
Using time varying boundary conditions (ground water levels) from the regional model (step 3) a more detailed model simulation is carried out using the local ground water model for the reservoir area. This model uses detailed information on the sediment layer provided by step 2.

Main output: Detailed three-dimensional ground water flow regime including recharge of water from the reservoir to the ground water.

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Step 5) MIKE SHE/MIKE 11 local model for the river branch system on the Slovak floodplain.
Using time varying boundary conditions from the regional ground water model (step 3), a detailed ground water/surface water flow simulation for the river branch system is carried out. The output from this model forms the basis for the description of the ecological conditions in the river branch system.

Main output: Moisture content in the unsaturated zone, ground water levels, infiltration to the ground water and seepage to the Old Danube, depth and areal extent of inundations of flood plains.

Step 6) DAISY crop growth and nitrate leaching
Using time varying ground water levels from the regional ground water model (step 3) DAISY unsaturated flow, crop growth and nitrate leaching simulations are carried out.

Main output: Crop development parameters (leaf area index and root depth), crop yield, nitrate leaching and irrigation requirements distributed in time and space.

Step 7) MIKE SHE geochemical modelling
Based on the flow field from the local reservoir model geochemical modelling is carried out. Results from the sediment transport modelling and results from eutrophication and water quality modelling (step 2) are used to estimate the concentration of various species (oxygen, nitrate, organic matter etc.) in the water that recharges the ground water from the reservoir and the Danube, respectively. Nitrate leaching simulated with DAISY (step 6) are used to estimate nitrate concentrations in the water that percolating to ground water in the remaining parts of the model area.

Main output: Concentration of various species (nitrate, nitrite etc) in the ground water distributed in time and space.

3. Modelling studies in the Danubian Lowland - a few results

Comprehensive modelling studies have been carried out under the PHARE project (Slovak Ministry of Environment, 1995). In the present paper a few selected results are presented with regard to hydrology of the floodplains.

3.1. MODEL CONSTRUCTION FOR RIVER BRANCH SYSTEM

The complexity of the floodplain with its river branch system is indicated in Figs. 4 and 5 for the 20 km reach downstream the reservoir on the Slovakian side where alluvial forest occurs. In order to enable predictions of possible changes in floodplain ecology it is crucial to provide a detailed description of both the surface water and the groundwater systems in this area as well as of their interaction. For this purpose the MIKE SHE-MIKE 11 coupling is required.

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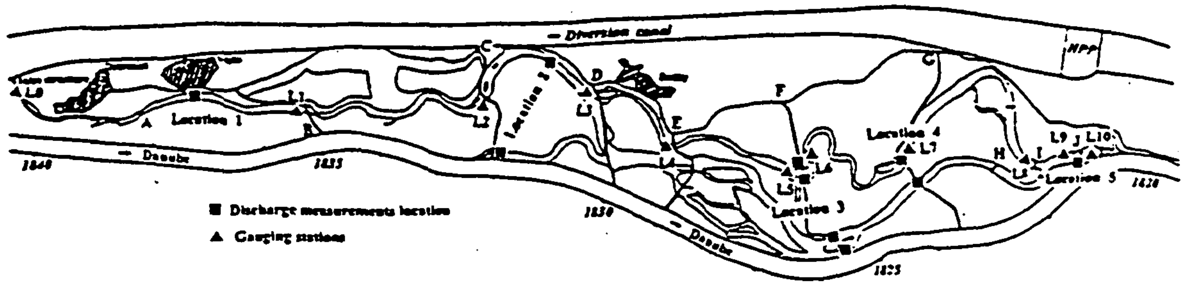

Figure 4. Layout of the river branch system on the Slovakian side of the Danube.

Before the damming of the Danube in 1992 the river branches were connected with the Danube during periods with discharge above average. However, some of the branches were only active during flood situations a few days per year. After the damming the water level in the Old Danube has decreased significantly. Therefore, in order to avoid that most water drained from the river branches to the Old Danube, resulting in totally dry river branches, the connections between the Danube and the river branches have been blocked except for the downstream one at chainage 1820 rkm (see Fig. 4). Instead, the river branch system receives water from an inlet structure in the hydropower canal at Dobrohost (see Fig. 4). This weir has a design capacity of 234 m^3/s and together with the various hydraulic structures in the river branches, it controls the hydraulic, hydrological and ecological regime in the river branches and on the floodplains. The extent of the floodplain model area is indicated in Fig. 1. and a perspective view of the area with the river branch system and floodplains are shown in Fig. 5. The horizontal discretization of the model is 100 m, while the groundwater zone is represented by two layers. Several hundreds of cross-sections and more than 50 hydraulic structures in the river branch system were included in the MIKE 11 setup for the river system.

For the pre-dam model setup the surface water boundary conditions comprise a discharge time series at Bratislava and a water level - discharge relationship at the downstream end (Komarno). For the post-dam model setup the Bratislava discharge time series has been divided into three discharge boundary conditions, namely at Dobrohost (intake from hydropower canal to river branch system), at the inlet to the hydropower canal and at the inlet to Old Danube from the reservoir. For the groundwater system, time varying ground water levels simulated with the regional ground water models act as boundary conditions. The Old Danube river forms an important natural boundary for the area. The Old Danube is included in the model, located on the model boundary, and symmetric ground water flow is assumed under the river. Hence, a zero-flux boundary condition is used as boundary condition for ground water flow below the river.

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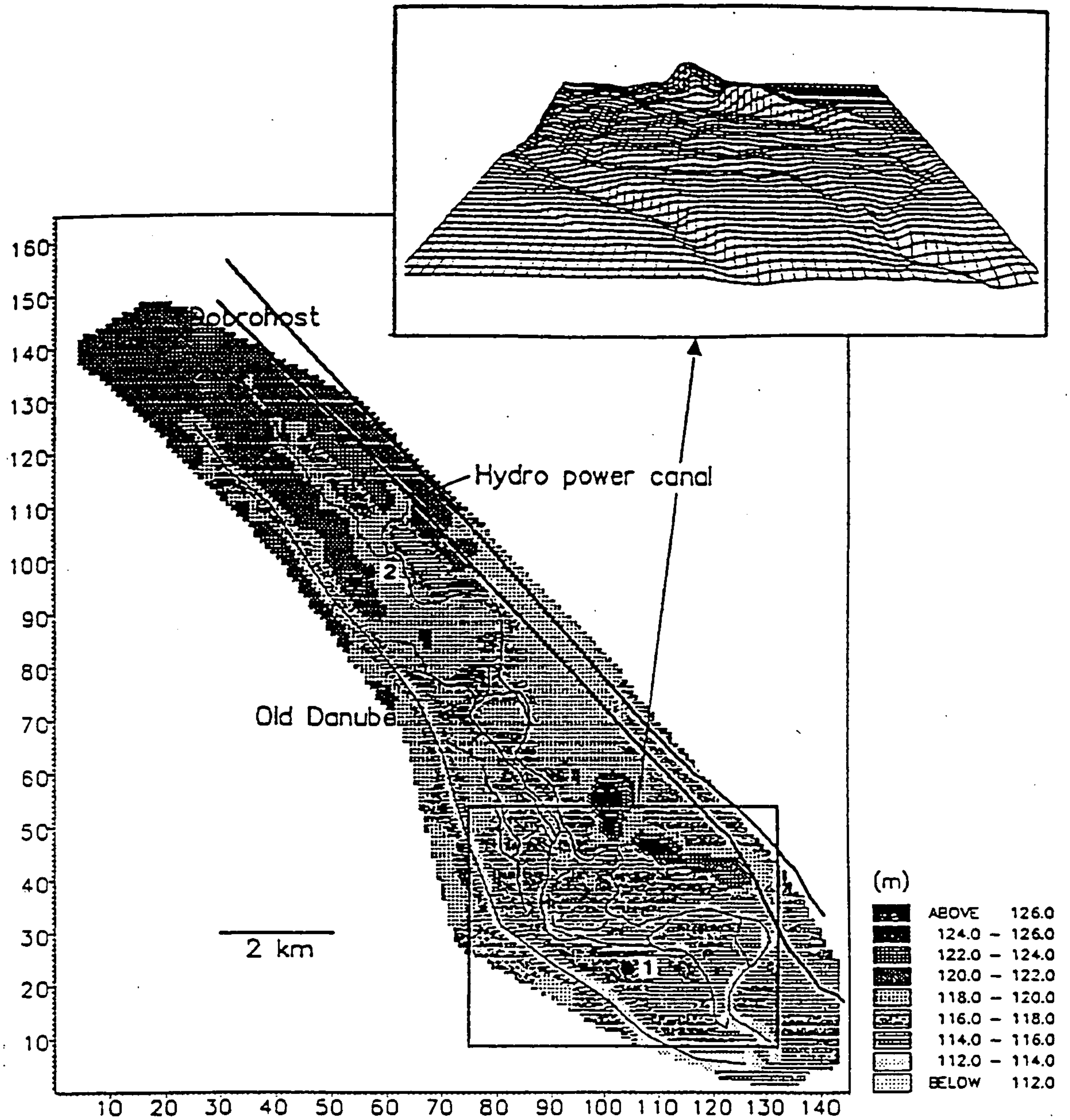

Figure 5. Plan and perspective view of the surface topography, of the river branches and the related flood plains as represented in a model network of 100 m grid squares.

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3.2. MODEL CALIBRATION AND VALIDATION

The hydraulic model of the river branch system was calibrated against water level and discharge data measured during a three-week field campaign in June 1994 (Holubova et al., 1994). It appeared that the major part (up to 80%) of the inlet discharge at Dobrohost disappears between the intake structure and the downstream confluence with the Old Danube. The reason for this water loss is infiltration from the river branches to the aquifer system from where a significant part of it discharges the Old Danube which has a lower water table than the river branch channels. The calibration parameters included Manning numbers in the channels, flow capacity for some of the hydraulic structures and leakage coefficients for the channel beds. Although the structure geometry were known from measurements and design the field conditions were some times different due to blocking of culverts by dead trees etc.

The model was validated by testing its ability to reproduce water levels measured during the summer of 1993. Some of the validation results are shown in Fig. 6. It is seen from this figure that the largest deviations between model simulations and observed water levels are at the upstream location (L1). The reason for this is that the model parameter values describing the hydraulic structures correspond to the situation during the calibration period, i.e. 1994, while some modifications, mainly of crest elevations, were made to a few of the hydraulic structures in the upstream part of the river branch system between the 1993 validation period and 1994 . Thus the model setup corresponds to the new crest parameters, while the field data during the 1993 validation period correspond to old ones. Nevertheless, in spite of these minor inconsistencies between model parameters and field situation the simulation results correspond well to the measured water levels both with regard to dynamics and levels.

3.3. MODELLING OF FLOODPLAIN DYNAMICS

To illustrate the complex functioning of the MIKE SHE - MIKE 11 floodplain model and the interaction between the surface and subsurface hydrological processes model simulations for a period in June - July 1993 are shown in Figs. 7 and 8.

Fig. 7 presents the inlet discharges at the upstream point of the river branch system (Dobrohost), while the discharges and water levels at the confluence between the Danube and the hydropower outlet canal downstream of Gabcikovo during the same period are shown in Fig. 8. Fig. 7 shows furthermore the soil moisture conditions for the upper two m below terrain and the water depth on the surface at location 2. Similar information is shown for location 1 in Fig. 8. A soil water content above 0.40 (40 vol. %) implies that the soil is saturated. Location 2 is situated in the upstream part of the river branch system, while location 1 is located in the downstream part (see Fig. 5).

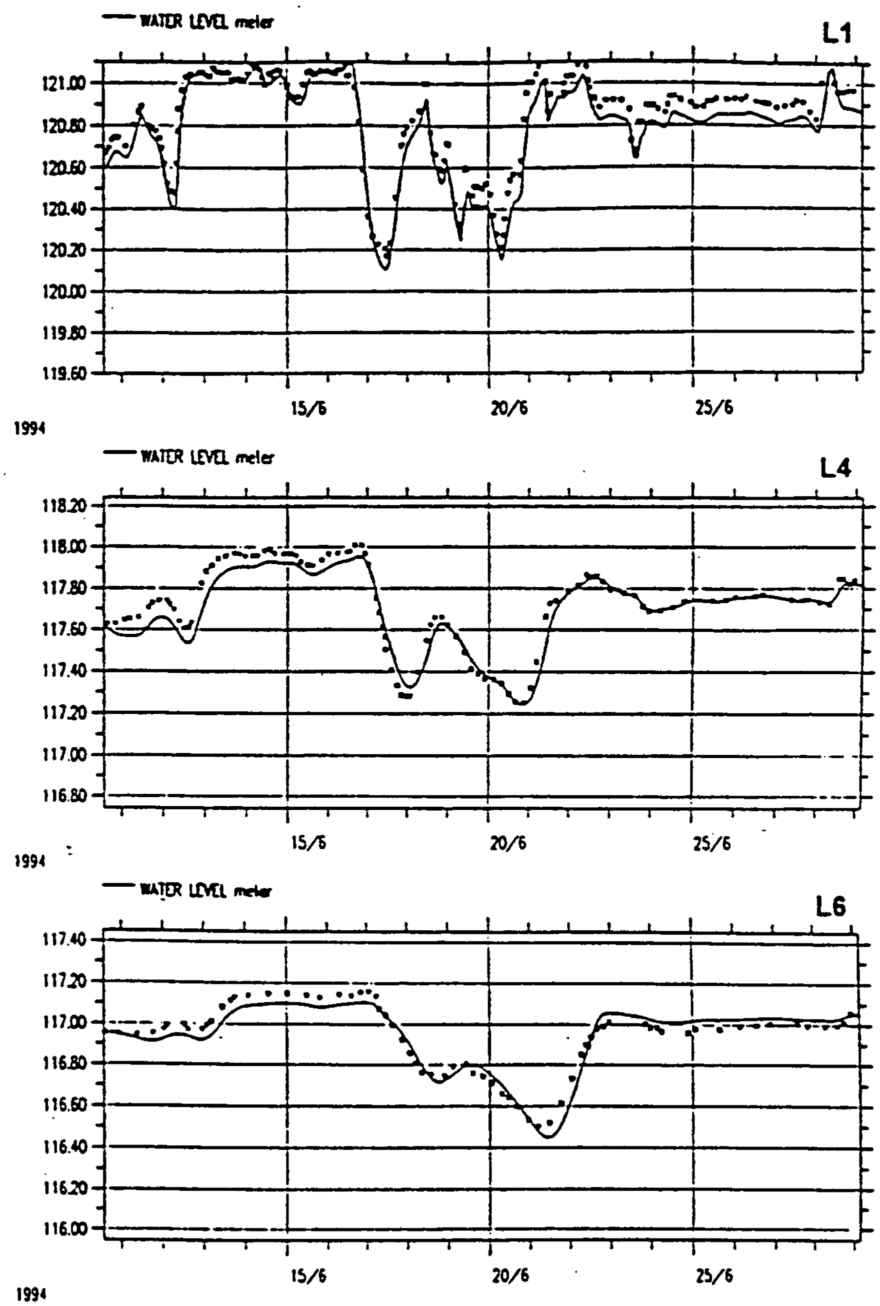

Figure 6. Validation of the river branch model against measured water levels for July-August 1993. The locations of the three sites L1, L4 and L6 can be seen in Fig. 4.

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At location 2 (Fig. 7) flooding is seen to occur as a result of river spilling (surface inundation occurs *before* ground water table rises to surface) whenever the inlet discharge exceeds approximately 60 m^3/s. The soil moisture content is seen to react relatively fast to the flooding and the soil column becomes saturated. In contrary, full saturation and inundation does not occur in connection with the flood in the Danube in July, but the event is seen in terms of increasing groundwater levels following the temporal pattern of the Danube flood.

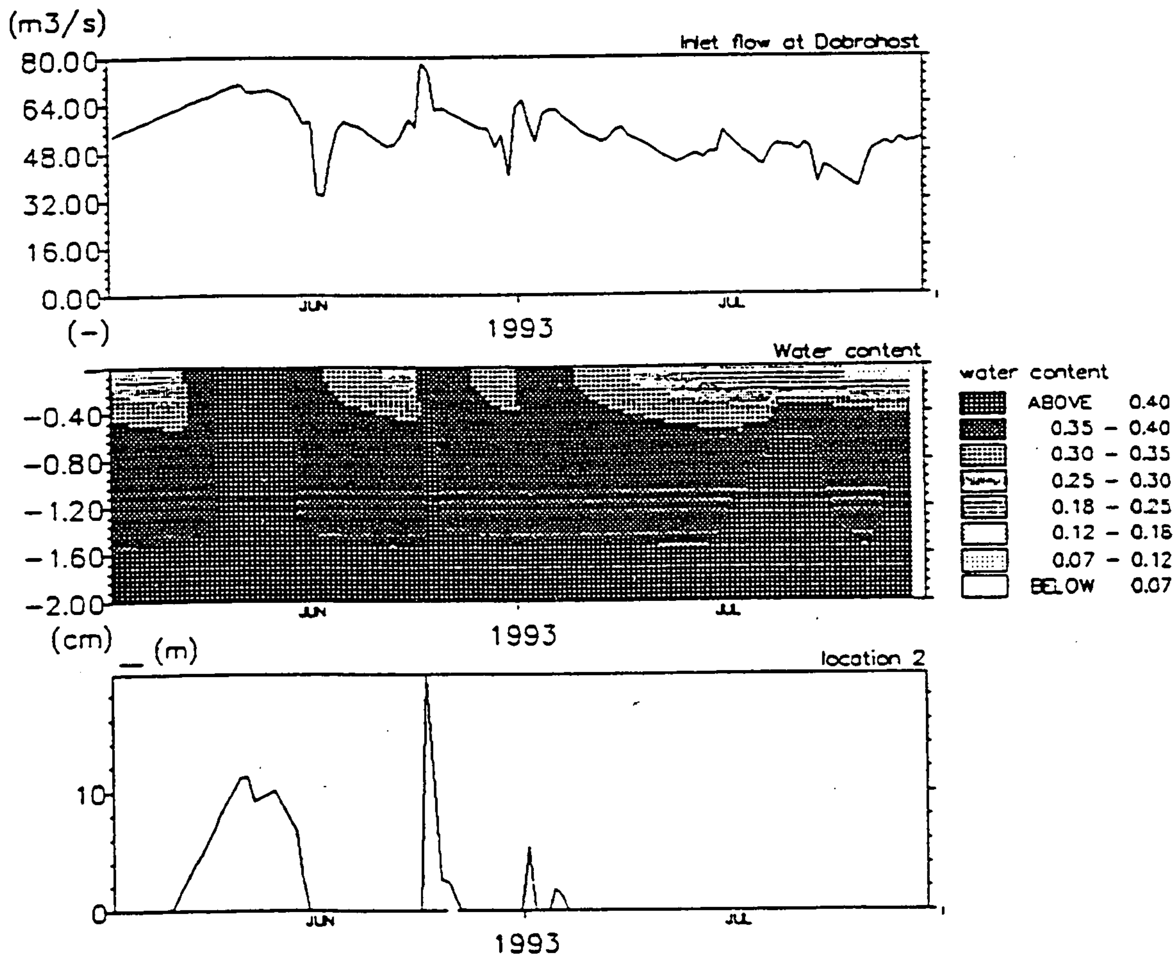

Figure 7. Observed inlet discharge to the river branch system at Dobrohost; simulated moisture contents at the upper two m of the soil profile at location 2 and simulated depths of inundation at location 2 during June-July 1993.

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At location 1 (Fig. 8) the conditions are somewhat different. During the simulation period location 1 never becomes inundated due to high inlet flows at Dobrohost. However, during the July flood in Danube inundation at location 1 occurs as a result of increased ground water table caused by higher water levels in river branches due to backwater effects from the Danube. The surface elevation at location 1 is 116.4 m which is 0.4 m below the flood water level shown in Fig. 8 at the confluence (5 km downstream of location 1). It is noticed that the inundation at this location occurs as a result of ground water table rise and not due to spilling of the river (surface inundation occurs *after* the ground water table has reached ground surface).

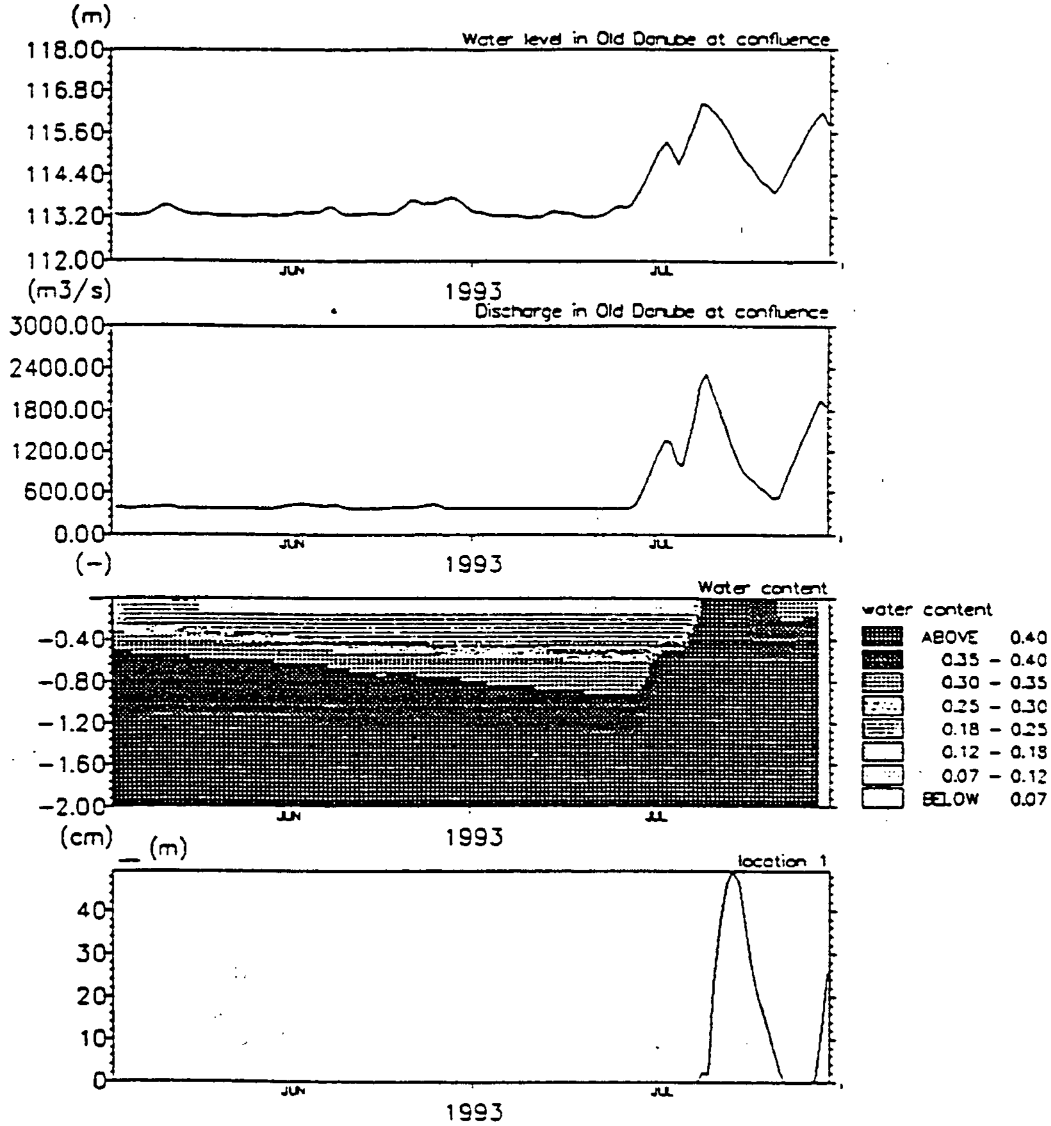

Figure 8. Simulated discharge and water levels in the Danube at the confluence between Old Danube and the outlet canal from the hydropower plant; simulated moisture contents at the upper two m of the soil profile at location 1 and simulated depths of inundation at location 1 in the river branch system during June-July 1993.

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3.4. RESULTS OF MODEL APPLICATION

The floodplain model is a management tool which can simulate the operation of the hydraulic structures, enabling an optimization of the hydraulic and ecological conditions for the unique floodplain environment. The floodplain model provides detailed information in time and space about water levels in river branches and on the floodplains, groundwater levels and soil moisture conditions in the unsaturated zone. Such information can directly be compared with quantitatively formulated ecological criteria.

As an example of the results which can be obtained by the floodplain model Fig. 9 shows a characterization of the area according to flooding and depths to groundwater. The map has been processed on the basis of simulations for 1988 for pre-dam conditions. The classes with different ground water depths and flooding have been determined from ecological considerations according to requirements of (semi)terrestrial (floodplain) ecotopes. From the figure the contacts between the main Danube river and the river branch system is clearly seen. Similar computations have been made by alternative water management schemes after damming of the Danube. The results of one of the hypothetical post-dam water management regimes, characterized by average water flows in the power canal, Old Danube and river branch system intake of 1470 m^3/s, 400 m^3/s and 45 m^3/s, respectively, are shown in Fig. 10. By comparing Fig. 9 and Fig. 10 the differences in hydrological conditions can clearly be seen. From such changes in hydrological conditions inferences can be made on possible changes in the floodplain ecosystem.

Further scenarios (not shown here) have, amongst others, investigated the effects of establishing some underwater weirs in the Old Danube and in this way improve the connectivity between the Old Danube and the river branch system.

4. Conclusions

The ecological system of the Danubian Lowland is so complex with so many interactions between the surface and the subsurface water regimes and between physical, chemical and biological changes that a comprehensive mathematical modelling system of the distributed physically-based type is required in order to provide quantitative assessments of environmental impacts.

Such modelling system coupled with a comprehensive data base/GIS system has been developed. The integrated system makes it possible at a quite detailed level to make quantitative predictions of the surface and ground water regime in the floodplain area, including e.g. frequency, magnitude and duration of inundations. Such information constitutes a necessary basis for subsequent analysis of flora and fauna in the floodplain.

In the present chapter some of the capabilities of the modelling system have been illustrated by a few selected results on flood plain hydrology.

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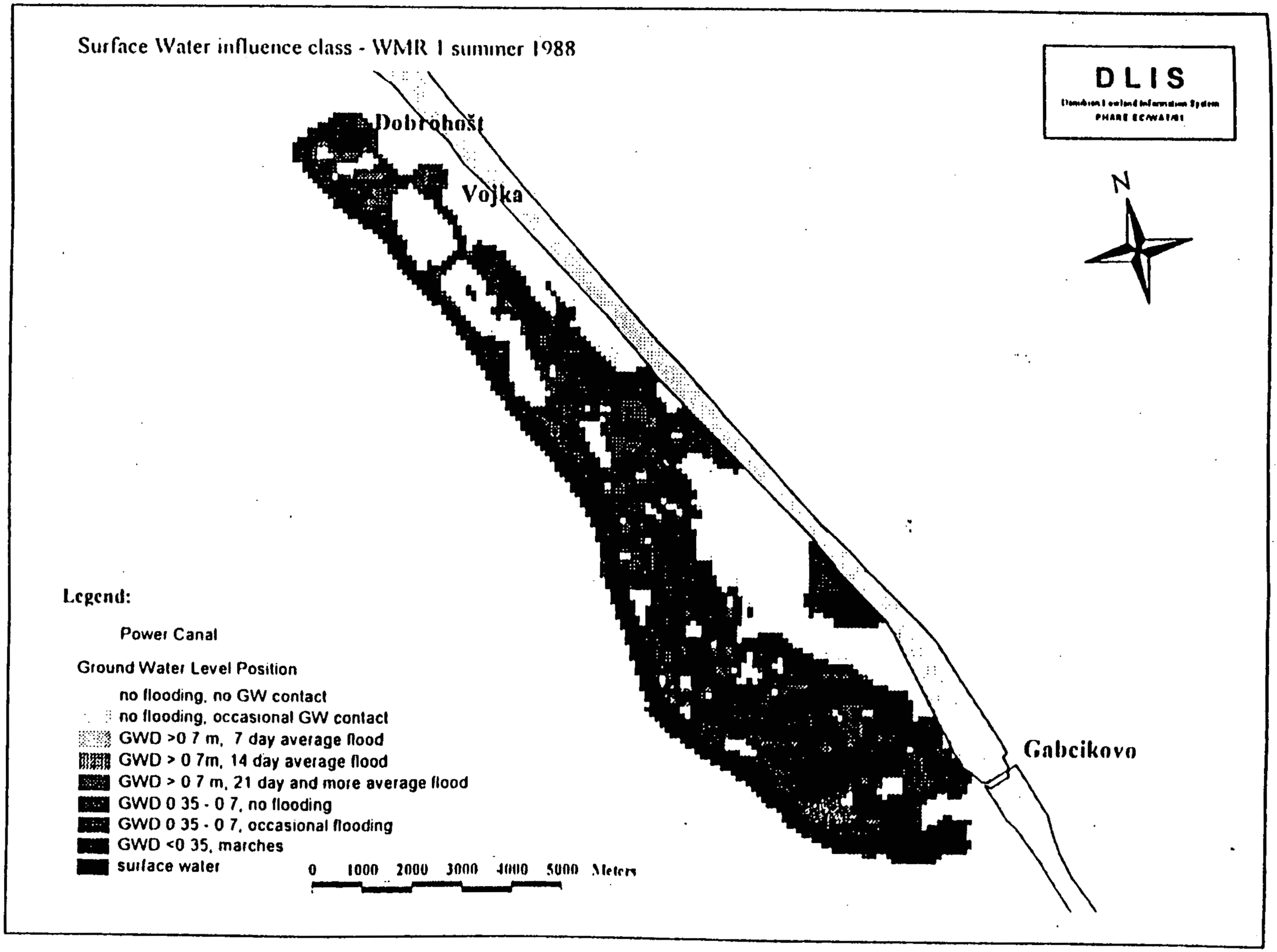

Figure 9. Hydrological regime in the river branch area for 1988 for pre-dam conditions characterized in ecological classes.

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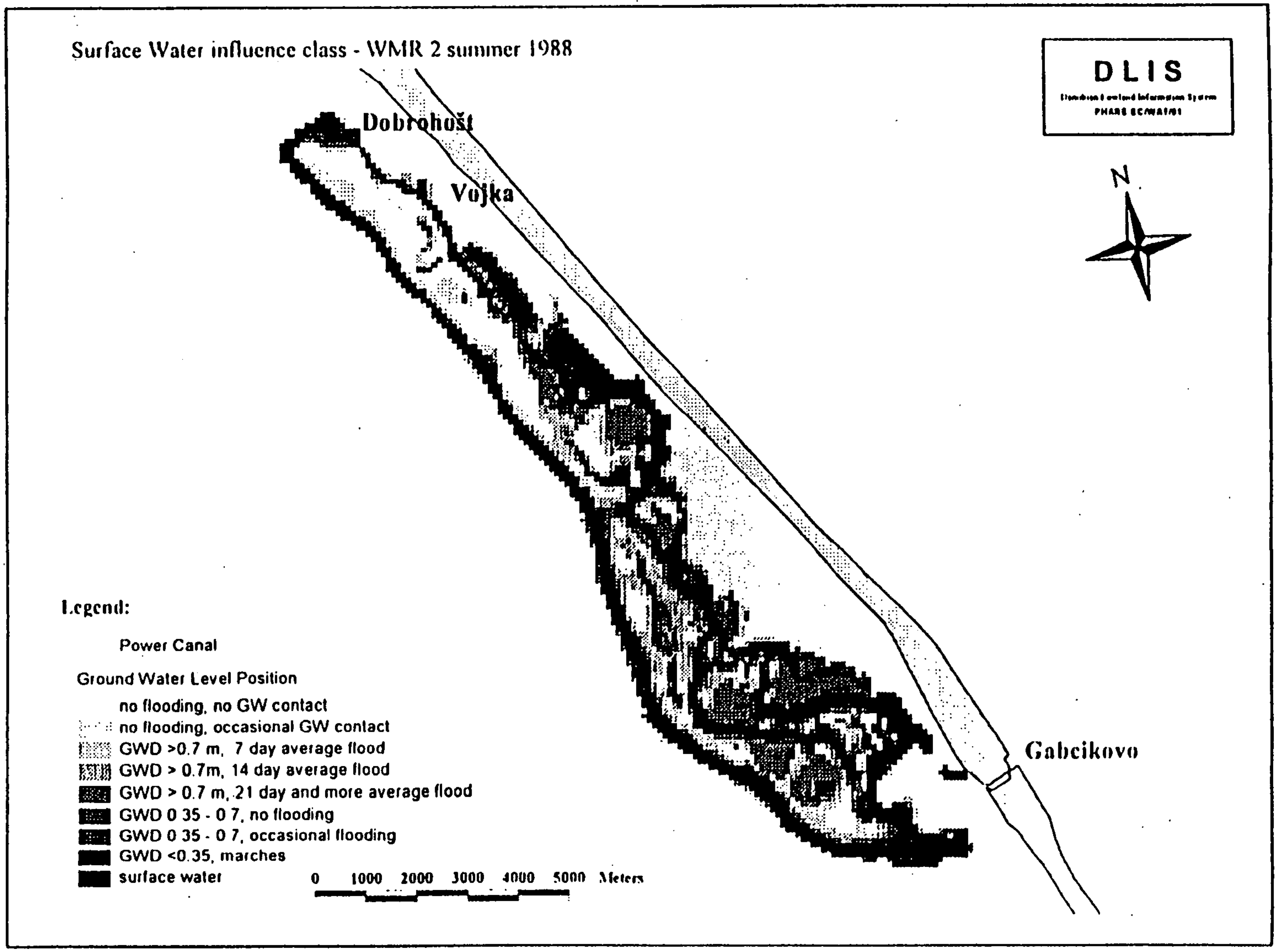

Figure 10. Hydrological regime in the river branch area for a post-dam water management regime characterized in ecological classes. The scenario has been simulated using 1998 observed upstream discharge data and a given hypothetical operation of the hydraulic structures.

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5. Acknowledgement

The PHARE project was executed by the Slovak Ministry of the Environment and supported financially by the European Commission. A Danish-Dutch consortium of six organizations was selected as Consultant for the project. The Consultant was headed by Danish Hydraulic Institute (DHI) and comprised the following associated partners: DHV Consultants BV, The Netherlands; TNO-Applied Institute of Geoscience, The Netherlands; Water Quality Institute (VKI), Denmark; I Krüger Consult AS, Denmark; and the Royal Veterinary and Agricultural University, Denmark.

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CHAPTER 13B
COMMENT ON 'A DISCUSSION OF DISTRIBUTED HYDROLOGICAL MODELLING' BY K. BEVEN

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1. Introduction: Terminology and Content of Comment

Before, commenting on Keith Beven's questions we should like to emphasize two issues relating to our terminology and our assessments of the key objectives of distributed hydrological modelling. Indeed, since the terminology used by Beven differs in some respects significantly from the terminology used in some of the other chapters of this book, we should like first to define some key terms as we have used them in the present discussion. We begin by distinguishing between a model and a modelling system. A *model* is defined as a particular hydrological model established for a particular catchment. A *modelling system* or a *model code*, on the other hand, is defined as a generalized software package which can be used without program changes to establish a model with a range of generic basic types of equations (but allowing different parameter values) for different catchments. We then define *model validation* as the validation of a site-specific model, while *code verification* refers to the testing of algorithms etc. Our terminology is defined and discussed in more details in this book by Refsgaard (Chapter 2) and Refsgaard and Storm (Chapter 3).

With respect to the objectives of distributed hydrological modelling we see four different major types of objectives, namely:

(a) Simulation of discharges under stationary catchment conditions.
(b) Simulation of the effects of catchment changes due to human interference, such as land use change, groundwater development and irrigation.
(c) Water quality and soil erosion modelling.
(d) Research on hydrological processes.

Whereas objective (a) can be equally well addressed by simpler hydrological models, such as the lumped conceptual models, we see no real alternatives to distributed models with respect to objectives (b) and (c). Furthermore, distributed physically-based models are the most suitable for many research purposes, since they directly allow the user to incorporate new hypotheses on process descriptions for testing. Most of the criticisms made during the past decade against distributed models as being unnecessarily complex, have been made in relation to discharge simulations (objective (a)). Hence, although we agree with Beven as to many of the fundamental problems outlined in Chapter 13A, we

M. B. Abbott and J. C. Refsgaard (eds.), Distributed Hydrological Modelling, 279–287.

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believe it is important to realize that for many important modelling purposes there are at present no realistic alternatives to the distributed physically-based approach.

With our background as being involved both in engineering hydrology (consultancy) and in research it might be expected that the engineering part of our activities should require simple approaches (which is common in consulting work). However, our experience it that, although we have the full range of model codes available in-house, including the simple traditional ones, many of the environmental projects, in which we have been involved in fact required distributed modelling. Thus, we have as modellers chosen to use distributed catchment models because we believed that these tools, in spite of the scientific simplifications, errors and limitations, can provide the best possible information to support decisions in water resources management.

2. The basic problem with distributed hydrological modelling

Beven argues that the currently available physically-based distributed model codes are in fact lumped conceptual model codes. The justifications for this statement are the lack of physical realism of certain process descriptions such as those of overland flow and preferential flow paths in the unsaturated zone.

To our mind, however, there are still clear fundamental differences in the way a typical lumped conceptual code, such as the Sacramento, and a typical distributed physically-based code, such as the MIKE SHE, operate with regard to process descriptions, spatial variability of hydrological variables, physical realism of parameter values, and applicability. As examples of models which cannot, meaningfully, be classified as lumped conceptual, consider the geochemical model in Engesgaard (Chapter 4) or the coupled MIKE SHE - MIKE 11 floodplain model in Sørensen et al. (Chapter 12). To state that there are no differences between distributed physically-based and lumped conceptual model types adds more to the confusion than it does to the clarification of these issues.

We agree that in many applications the distributed physically-based codes have been used to construct hydrological site-specific models that may rather be classified as lumped conceptual, albeit very detailed ones. This can for example be argued with respect to a SHE application to catchments in India, where grid sizes of 1- 4 km were used (Refsgaard et al., 1992). However, this does not imply that the code in general cannot be used to construct distributed physically-based models.

We acknowledge that the present knowledge on hydrological processes suggests the need to construct much more complex hydrological models than were foreseen in the blueprint for a physically-based model of Freeze and Harlan (1969), and this process is already well under way. The very complex soil erosion processes described by Lørup and Styczen (Chapter 6) is a good example in this respect. However, this does not make the need for such models less; it 'just' makes it more difficult to establish a generally applicable code. We regard this development of improved hydrological codes as a continuous, on-going and apparently never-ending, process, whereby new knowledge about processes and new capabilities of accommodating new data types have regularly

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to be added. Indeed, from this point of view one of the chief merits of the development of such codes is that they provide a framework for the accumulation and integration of new knowledge, introduced from many different disciplines by a very considerable number of individual experts. A modelling system of this kind then also provides the technical means for *consensus building* across the widest possible range of disciplines in hydrology.

Thus, although we agree to a large extent with Beven's summary statement: "it cannot be assured that distributed models are based on the correct equations to describe hydrological reality at the grid scale", we consider it an expression of a rather pessimistic view. What is 'correct' and what is 'reality', and how 'correctly' do we have to describe 'reality' in different situations? (For the definition of such terms as 'reality' and 'truth' as used in hydroinformatics, see Abbott, 1994). Certainly, one can always specify performance criteria that are so strict that no model will ever pass a validation test. We consider it a more fruitful challenge to hydrological modelling to develop model codes and construct models on the basis of the best available knowledge, knowing that this basis can be improved in the future, and to use this at any time as the best *current* basis for decision making.

Furthermore, Beven argues that "while they (the distributed models) are overparameterised for the purposes of estimating discharges, they have not been properly tested in terms of simulating the internal state variables". While we agree that this may often be the case in some particular applications, we do not see how Beven can make such induction from a few examples to a universal statement. The important fact for us is that distributed physically-based codes enable model validation against internal state variables, and furthermore allow rigorous and transparent parameterisation schemes to be used. Whether models are used in such a responsible way or rather everything is confused and unjustified claims are made is the responsibility of the individual model users and should not be confused with the applicability of and general statements about the codes.

3. Examples of Successfully Validated Distributed Models at the Catchment Scale

In accordance with the terminology defined in Section 1, we associate model validation with site-specific models. A generic model code can be verified, implying the successful testing of the mathematical algorithms and the program code. Furthermore, a given model validation should always be related to pre-specified performance criteria. Within this framework, many examples of successful model validations exist, although the performance criteria most often have not been pre-specified explicitly. Refsgaard (1996) reports successful validations of lumped and distributed models from a study on Zimbabwean catchments, where rigorous test schemes were used. Jensen and Jørgensen (1988) describes a successful post-audit study of the distributed groundwater/surface water model for the 1000 km^2 Danish Suså catchment, developed and calibrated 10 years earlier (Refsgaard and Hansen, 1982). Originally, the Suså model was calibrated against soil moisture data from four plots, groundwater heads from about 40

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observation wells and discharges data from 6 stations. During the post-audit period. 1981-87, the groundwater abstractions were slightly different from those in the original study, 1951-80. The post-audit comprised validation against data from 38 groundwater observation wells and 4 discharges stations, and the model predictions were found to match the values to the same degree of accuracy as during the calibration period.

In Beven's terminology, model validation refers to a kind of general validity of the model code. We agree that a model code can, in principle, never be documented as universally valid. Each model application, with its inherent assumptions and assessments made in connection with construction and possibly calibration of the model, must be validated separately. Because one model user succeeds in constructing and validating a model for given conditions in the case of one catchment, this does not imply that another user can automatically construct a valid model for another catchment using the same generic code.

The question of model validation applies equally well to all model types. Thus no lumped conceptual model code can be claimed to be universally valid either. Instead of the validity of a model code, we talk about an apparently less strict term: credibility of a given model code. By this we mean that a generic code, which has been used to construct many successfully-validated models covering a wide range of specific types of application gains a higher credibility than another code which has not been through such a chain of experience accumulation through incremental development.

In our opinion, the issues of model validation and code verification deserve much more effort in the future, and there is certainly a strong need for a common terminology (see Refsgaard, Chapter 2; Refsgaard and Storm, Chapter 3).

4. Are improved Process Parameterisations possible ?

Beven argues that the two main problems related to process descriptions (parameterisations) are (1): "the problem of change of scale and heterogeneity of parameters, even if the small scale equations were correct at the local scale" and (2): "the small scale equations may not be correct at the local (profile to plot) scale". We agree to this general statement as well as to the examples given by Beven. In fact, on the basis of our past 20 years experience with SHE and MIKE SHE we agree to the fundamental dilemma in distributed physically-based modelling outlined by Beven, namely that when limitations of the descriptive equations are recognised, it is necessary to introduce more complex parameterisations with (often) more parameters, and these may not always be easily measured and may also introduce new problems of heterogeneity.

Again, however, although we agree to Beven's problem assessment, our conclusions with regard to the modelling strategy that needs to be followed in order to accommodate these problems are quite different.

From a research point of view, we do not necessarily consider it a problem, but rather an inconvenience that more knowledge about processes and a greater access to field data lead to more complex models. If the objective of research is to achieve

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improved understanding of the hydrological processes in nature, new knowledge will inevitably have to be incorporated in models in order to imitate nature and test new theories. If we take the route of working with simpler models, we would not be able to utilize all available information on hydrological data and, not knowing beforehand which new data contains significant information for the problem at study, we would risk stopping further progress in the deeper and more detailed understanding of hydrological processes.

From a practical model application point of view, we can agree with Beven that if the sole objective is to model the rainfall-runoff process and predict discharges at the outlet of a catchment, then simpler models are adequate. According to our experience from several studies, such as the intercomparative study (Refsgaard, 1996) using a lumped conceptual code (NAM), a distributed physically-based code (MIKE SHE) and a semi-distributed conceptual/physically-based code (WATBAL) on Zimbabwean catchments, lumped conceptual and semi-distributed models with relatively simple process descriptions are generally just as good as more complex models such as MIKE SHE.

However, as outlined above there are many modelling objectives, such as prediction of effects of land use changes or ground water abstractions, and simulation of water quality and soil erosion, for which we see no alternatives to even more complex model codes than the existing ones.

5. Disaggregation and Scale Problems

We agree that the aggregation and scale problems are very fundamental, and indeed are of a fundamental nature. Thus, process equations and effective parameter values which are valid at one scale may not necessarily be valid at larger or smaller scales. The fact that data collection is carried out at a large range of scales does not make the problem easier. Thus, we agree to the need for identifying appropriate scaling procedures.

Comprehensive researches have been carried out in a stochastic framework for certain aspects of subsurface hydrology, such as groundwater transport (Gelhar, 1986) and unsaturated zone flow (Jensen and Mantoglou, 1992). A few attempts have also been made in the case of rainfall-runoff processes on hillslopes (Freeze, 1980). In recent years this issue has also received considerable attention in connection with simulations of the interaction between the land surface and the atmosphere (Entekhabi and Eagleson, 1989; Famiglietti and Wood, 1994). Much of the same kind of problem has been extensively studied in other fields, and notably in the theory of turbulence (e.g. Leslie and Quarini, 1979; Leonard, 1974), where the way in which in which it leads to the introduction of additional higher order terms in existing continuum equations has been rather fully analyzed.

All the reported studies on scaling in hydrology, however, have highlighted the problem only for particular cases and have, at best, provided theoretically scaling methodologies for such cases. Thus, no universal methodology has yet been developed, nor appears to be within sight in the short term.

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In addition to the above research areas, which can be characterized as local scale problems, we often encounter scale problems when applying distributed models for several hundred km^2 size catchments (see Storm and Refsgaard, Chapter 4). In engineering applications of distributed models, the scale problems are, in practice, often circumvented through specific model calibration. In this way parameter values are indirectly fitted to the particular scale, and it is fully accepted that these may not necessarily be valid at other scales. Thus, if models are not calibrated, or, in particular, if models after calibration and validation are used with different spatial discretizations, then the use of appropriate scaling procedures becomes important.

6. Assessment of Uncertainty of Model Predictions

Beven emphasizes the importance of associating model predictions with estimates of predictive uncertainty. We agree that this is a very important issue which requires more attention in the future.

The most common method today of assessing model prediction uncertainties is sensitivity analysis, but this is most often carried out in a rather qualitative, unsystematic manner. More comprehensive and rigorous methodologies require a joint stochastic-deterministic modelling approach, such as state space formulations or Monte Carlo techniques. As demonstrated for the Sacramento model code (Kitanidis and Bras, 1978) lumped conceptual models can easily be reformulated in a state space form and imbedded in a Kalman filtering framework enabling predictions of uncertainty bands to be placed upon discharges. Refsgaard et al. (1983) and Storm et al. (1988), using the NAM lumped conceptual code in a Kalman filtering framework, made analyses of the propagation of uncertainties due to both the uncertainty of rainfall input and the uncertainty of certain key model parameters. Joint stochastic-deterministic modelling, including the prediction of uncertainty bands due to incomplete knowledge of the spatial variability of hydraulic parameter values, is also common in subsurface hydrological modelling (Gelhar, 1986; Kros et al., 1993; Zhang et al., 1993; Jensen and Mantoglou, 1994).

Although joint stochastic-deterministic modelling is not yet common in distributed hydrological modelling, we agree with Beven that it is feasible, at least in the form of a Monte Carlo approach. Assessments of model prediction uncertainties will be useful in connection with the following types of applications of distributed models:

* Use of model results for supporting management decisions.
* Updating (data assimilation) by use of point data from traditional monitoring networks and spatial data from remote sensing.
* Inverse modelling in much more general contexts.

Among the methodologies reported in literature the Generalised Likelihood Uncertainty Estimation (GLUE) methodology appears so far as the most comprehensive approach, and one which we look very much forward to see further developed and applied.

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7. The Future of Distributed Modelling: on the Value of Data

As discussed by Refsgaard and Abbott (Chapter 1) distributed models have in general had less data available than they could have used. Thus, we agree with Beven that hydrological science is awaiting the development of new measurement techniques, especially with respect to spatial data and their heterogeneity.

However, it appears to us that several developments in these years indicate that this situation is likely to become significantly improved within the coming years. Firstly, new data sources are emerging, such as remote sensing data from new active sensor systems and geophysical data from new sensor types as supplements to geological data, while new in-situ water quality sensors are rapidly being developed, and weather radar data are gradually becoming more reliable. Secondly, the widespread use of GIS and other data base systems are gradually making the existing data, which previously in practice were not accessible in large volumes, much more easily available and applicable.

In today's water resources management systems, data bases are being developed in many places and models are being used to some extent; but very seldom is all available data used on a routine basis together with distributed models. We foresee that distributed models in the future will be integrated with permanent comprehensive data collection systems and data bases as decision support tools in water resources management.

Beven argues that with more available data "the problem of overparameterisation is consequently greater". As discussed by Refsgaard and Storm (Chapter 3) and Storm and Refsgaard (Chapters 4) we believe that this is not necessarily correct. A key issue in this respect is how the parameterisation is carried out. We advocate avoiding make too many degrees of freedom in connection with calibration procedure. Hence, almost all distributed data should be data which are not subject to calibration.

8. Conclusions

Basically, we agree with Keith Beven's listing of the problems existing in the present generation of distributed physically-based model codes with respect to general (in)adequacy of local scale process descriptions, heterogeneity and scaling problems, and the need to make assessments of uncertainties in model predictions. Furthermore, we agree that many non-documented claims have been made about the capabilities and overall validity of generic model codes and, particular, individual models. We agree that these issues deserve much further attention in terms of fundamental and applied research.

However, although we basically agree with Beven's assessment of these problem areas, our conclusions with regard to the future of distributed hydrological modelling are in most respects quite different.

Whereas we agree that for runoff prediction the complex distributed physically-based models are generally unnecessary, we disagree with Beven's more general conclusion that there is no benefit in using the comprehensive distributed physically-based codes. and that distributed models should therefore be made simpler. In our view the main justification for the distributed physically-based codes are the demands for prediction of effects of such human intervention as land use change, groundwater abstractions, wetland management, irrigation and drainage and climate change as well as for subsequent simulations of water quality and soil erosion. For these important purposes, we see no alternative to further enhancements of the distributed physically-based model codes, and we believe that the necessary codes in this respect will be much more comprehensive and complex than the presently existing ones.

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CHAPTER 14
HYDROLOGICAL MODELLING IN A HYDROINFORMATICS CONTEXT

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1 Introduction to Hydroinformatics

The informational revolution of the last 30 years has fundamentally altered the traditional planning, modelling and decision-making methodologies of the water-related sciences and technologies. Information technology (IT) now plays an essential role in the sustainable development of water resources and the responsible management of the aquatic environment. The general availability of sophisticated computers with ever-expanding capabilities has given rise to an increasing complexity in terms of computational ability and in the storage, retrieval and manipulation of information flows. Hydroinformatics is the field of study of the flow of information and its processing by knowledge as applied to the flow of fluids and all that they transport.

From one point of view, hydroinformatics can be seen as having emerged from the well-established technology of computational hydraulics that utilises numerical modelling techniques to describe physical systems with sets of numbers and simulates the laws acting upon these systems with sets of operations on these numbers. The introduction of computational hydraulics some 20 years ago initiated a correspondingly significant revolution in classical hydraulics and a thorough reformulation of laws and concepts in order to accommodate the new possibilities represented by the discrete, sequential and recursive processes of digital computation. Now, with the introduction of hydroinformatics, more fundamental changes again are taking place.

In addition to water quantity and quality data, the information necessary to describe and assess the state of any given body of water must also include a plethora of social, legal and environmental factors. In this context, the typical information to be incorporated into a hydroinformatics system must include such variables as international and national laws, local bye-laws - either temporary or permanent - and any applicable physical, chemical and biological parameters. Added to this, flows of water, sediment, chemicals and other water-borne substances must be calculated and measured, and the sites and water quality parameters of the area's water users identified. Lastly, the locations and production rates of heat, chemical and biological pollution must be introduced into the representation, as well as the presence, position and capacity of control elements in the area, such as pumps, retention basins and treatment plants.

An important feature of a hydroinformatics system is that it allows the use of those numerical simulations that are subject to constraints expressed in natural language (such as applicable legislation, contracts and agreements). However, producing an accurate impact

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prediction requires a wealth of knowledge, much of which can only be obtained by studying previous experiences under similar circumstances. Hydroinformatics facilitates this assessment process by encapsulating expert knowledge and experience, and by making this knowledge available in informational form to hydro-scientists and engineers, thereby raising the level of their professional performance.

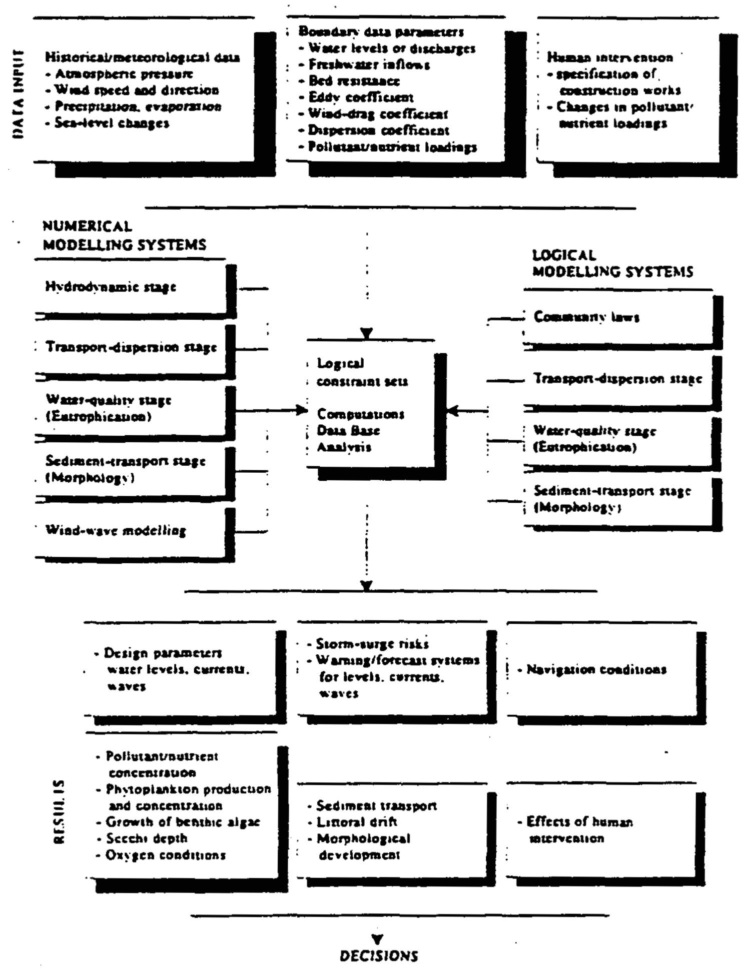

Figure 1 A prototype of a relatively complete hydroinformatics system for the Venice Lagoon, Italy.

In order to operate effectively, a hydroinformatics system may therefore be connected to measuring equipment, through a SCADA (supervisory control and data acquisition) system; it may contain numerical models to quantify the movements and changes within a body of water; it may use graphical interfaces to present the results of computations in a form which is understandable to a wide audience; it may assist in its own instantiations and in the interpretation of the results that it provides through the use of expert-advice systems; and it may store this information in data- and knowledge-bases. The size and complexity of such undertakings is well illustrated in Figure 1 (reproduced from Abbott, 1991, p. 33) that shows the schematisation of a relatively complete prototype for a hydroinformatics system of the Venice Lagoon in Italy as conceived already in 1989. Such systems have now entered service for the real-time control of urban drainage systems, they are being realised for coastal management systems and are being prepared for such applications as river-basin management and real-time control of irrigation systems.

The very development of hydroinformatics and the corresponding value that its integrating function adds to each of its components separately, leads in its turn to an accelerated development of measuring equipment, to much more sophisticated SCADA systems, to new modelling capabilities, to new means to relate measurements and models through data assimilation, automatic constitutive equation generation, automatic calibration procedures and other such applications of inverse and adjoint methods, to new data base technologies, to new user interfaces and indeed to any number of other such developments.

Despite the new ground that has already been broken by hydroinformatics, these systems are far from fulfilling their potential. Hydroinformatics research does not remain limited to the fields of hydraulics and hydrology alone, but has recourse to the latest IT developments in the fields of artificial intelligence (including machine learning, evolutionary algorithms and artificial neural networks), artificial life, cellular or finite-state automata and other, previously unrelated sciences and technologies.

Through studying and exploiting elements of these seemingly unrelated sciences, hydroinformatics is producing new and innovative solutions to hydraulic and hydrological problems, as represented by real-time control and diagnosis, real-time forecasting, calibration of numerical models, data analysis and parameter estimation. (*see* Verwey *et al*, 1994; IAHR, 1994; Babović, 1995). In particular, these new approaches can be used to generate important components of physically-based, distributed hydrological modelling systems by *inducing* models or sub-models of individual physical processes based only upon measured data. These (sub)models may then replace whole systems of complex, non-linear, differential equations that would otherwise require great skills from the modeller to calibrate and powerful computing devices to solve.

This chapter then introduces some research results with which the authors are familiar covering a range of applications varying from the modelling of certain aspects of the hydrological cycle to the analysis and control of complete water resources systems. The examples used are by no means exhaustive of the total range of applications of hydroinformatics in hydrology, but are indicative of the effectiveness of these new approaches in finding solutions to some long-standing problems in hydrological modelling.

2 Symbolic and Sub-symbolic Paradigms

In order to appreciate more fully the power of the new approaches described in this chapter, it is necessary to introduce a fundamental notion that expresses the essential difference between these approaches and the more traditional modelling approaches. This is the notion of the differentiation between symbols and signs and thus between symbol manipulation and sign manipulation. Symbols are an artefact of our beliefs about the natural world. These symbols are tokens that *stand in the place* of the objects that they represent. A collection of symbols, however, does not constitute a model. It is only when we interpret a collection of symbols, thereby giving them 'meaning' or semantic content, that we say that this collection of symbols becomes a *sign* that *points towards* a certain natural phenomenon. The set of symbols that we recognise as the one or the other of the Richards' equations for unsaturated flow then constitute a hydrological model because each such equation constitutes 'a collection of signs that serves as a sign' (Abbott, 1992). There can only be a finite number of signs in this world created by the modeller and the potential infinity of details in the physical-world that cannot be described within

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this limited sign vocabulary are often gathered together in the form of assumptions and simplifications that have to be applied in order to read a meaning into the sequence of symbols that is the differential equation. If the modeller accepts the limitations imposed upon the model by the assumptions and simplifications, then he or she accepts that this sign vocabulary, or language, is the best available description of the physical processes being considered. Natural systems, however, rarely conform to these assumptions. Subsequently, this symbolic language is commonly very restrictive for research into novel and innovative approaches.

The inherent limitations of the traditional modelling approach and its associated language are exemplified in the *Philosophical Investigations* of Wittgenstein (Part I, §§ 2-3):

> "Let us imagine a language ... (that is) ... meant to serve for communication between a builder A and an assistant B. A is building with building-stones: there are blocks, pillars, slabs and beams. B has to pass the stones, and that in the order in which A needs them. For this purpose they use a language consisting of the words 'block', 'pillar', 'slab' and 'beam'. A calls them out; - B brings the stone which he has *learnt* to bring at such-and-such a call. - Conceive this as a complete primitive language.
>
> "[On the other hand,] Augustine, we might say, does describe a system of communication; only not everything that we call language is this system. And one has to say this in many cases where the question arises 'Is this an appropriate description or not?' The answer is: 'Yes, it is appropriate, *but only for this narrowly circumscribed region, not for the whole of what you were claiming to describe*.' " (emphasis added)

In this simple example, the universe of discourse consists only of the words 'block', 'pillar', 'slab' and 'beam'. It would be impossible for the characters in this 'language game' ever to talk about 'doors', 'windows' or 'roofs' - let alone an entire house! Similarly, the hydrological modeller is restricted in his or her description of a hydrological catchment by the limited language of computational hydrology. The description of this catchment can only be as detailed as the model that is to be used to simulate the catchment processes. No amount of extra measured data will ever change the basic structure of the underlying differential equations of the model, but may only be used to adjust certain calibration parameters in order to bring the results of model simulations closer to the observed and measured phenomena. Since functional similarity to the natural system is supposed to be comprehended by the equations themselves, it is the calibration parameters that must then capture the correspondence between the model and the real world. These parameters serve in effect as *error compensation devices* that artificially adjust the model results to compensate for the fundamental discrepancies that exist between the real world and its differential equation representation in the model.

Calibration parameters are, however, usually not at all well-defined in nature. One may even ask 'What is the physical meaning of these parameters - how well are they grounded, and indeed are they grounded at all?'. We may indeed be able to read a certain 'physical meaning' into our calibration parameters, but they do not exist-as-such and are thus 'disconnected' in a fundamental way from the world that they are supposed to model.

The differential equations and the calibration parameters constitute the language of the hydrological modeller. The traditional approach to hydrological modelling is one of simply

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manipulating and adjusting these symbols in order to arrive at the best possible correspondence between model output and measured data. Nowadays we even recognise the process of symbol manipulation in the many commercial packages that claim to do just that (e.g. *Mathematica*. *Matlab*, etc.). Rarely is it possible for the modeller to create and incorporate new symbols and their associated signs into this language of discourse. The symbolic approach suffers from a rather thoroughly intractable problem of 'symbol grounding' (Harnad, 1990).

One of the greatest strengths of the new approaches described in this chapter is their ability to identify relationships and to induce models of measured data without requiring a detailed knowledge of physical hydrological characteristics *a priori*. One of the reasons for this is that many of these approaches manipulate the data at the level of the computer representation of the numbers. That is, the data are represented in our digital computers as *bits* and the operations upon this data then take place upon these individual bits. The modeller in this case has no direct influence upon the bits that convey the information. After translating the data that describe the symbols of our natural world into bit strings, the original symbols are then further irrelevant for the subsequent manipulations of bits. The algorithm operates at the level of the bits and is referred to as a sub-symbolic approach.

The computer is free to manipulate the bit strings, cutting them and rejoining them again at different places, flipping bits either randomly or in a controlled way. During this process the overall performance of the system can be observed and evaluated. This observation involves translating the bit information back into data that then acts once more as a sign which can be interpreted as a solution to a natural world phenomenon. This search for a solution takes place at the level of the bits and is unrestricted by the limitations of the language of our symbolic world. The computer itself, however, does not interpret the results of these recombinations at this sub-symbolic level.

The most important influence of the modeller in this process then is the translation or interpretation of the results being produced by the computer. These results should somehow 'make sense' to the modeller. The advantage of the sub-symbolic approach is that solutions may emerge whose signs point to other objects in the natural world that were not included in the original collection of symbols contained in the original model.

2.1 A SYMBOLIC APPROACH

It is possible to process strings of symbols which obey the rules of some formal system and which are interpreted by humans as ideas or concepts. It was the hope of early artificial intelligence researchers that all knowledge could be formalised in this way. However, the success of this approach relies very heavily upon selecting a sufficient set of symbols governed by a set of rules that is large enough to comprehend all possible conditions and this all linked together by some algorithm that is simple enough to program on the available, digital computing devices. It soon becomes obvious that even the simplest of tasks are extremely difficult to formalise in this way.

Expert or 'rule-based' systems represent one obvious example of a symbolic system. An expert system consists essentially of only two major components: a knowledge-representation component and an inference engine. Knowledge representation is the key component that 'translates' the utilities of the real-world into a finite collection of symbolic structures. In expert-systems this knowledge takes the form of rules. It is these rules that fundamentally discretise the

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world and reduce it to a finite number of configurations that are arranged in a tree, as exemplified in Figure 2.

```
IF    Water_level       > 10
AND   Discharge         > 25
THEN  Flooding_danger   = TRUE
```

```
IF    Flooding_danger  =   TRUE
AND   Discharge        <   40
AND   Gate_opening     <=  60%
THEN  Action           =   Open_gate_80%
```

```
IF    Flooding_danger  =  FALSE
THEN  Action           =  NONE
```

Figure 2 Knowledge representation in tree form in a simple expert system.

Given the represented knowledge in such a formalism, the task of the inference engine is then to find an instance that is most appropriate for a given situation. Correspondingly, the objectives of the inference engine are similar to those of any search problem. Most expert systems use either breadth-first or depth-first search strategies, as exemplified in Figures 3a and 3b, respectively. This figure illustrates the order in which an inference engine would 'visit' particular rules in a knowledge-base in order to locate the one that is most appropriate for a given input.

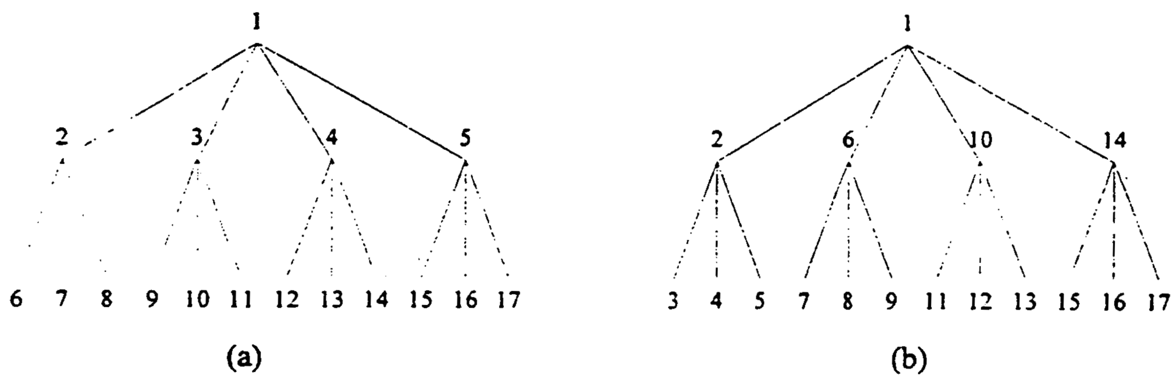

Figure 3 Search strategies in an expert system showing the order of searching the rules in the tree structure for (a) a breadth-first and (b) a depth-first search strategy.

From a cognitive point of view, it is argued that an expert system is the most appropriate programming environment for modelling logical human mental activities. It provides an automation of the reasoning process, with knowledge about the domain (knowledge base) clearly separated from the set of mental operations (inference engine) to be performed on the knowledge base. Babović (1991, pp. 12-16) described several features of expert systems that make them especially useful in complex systems. These features include:

- knowledge is presented in a highly declarative way. The symbolic appearance of the model can be interpreted by almost any user, allowing the logics of the model to be

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understood and permitting the alteration of the contents of the model in order to improve its performance;

- in addition to formal knowledge and established theories, fragmentary, ill-structured, approximate, incomplete, uncertain, heuristic and judgemental knowledge can be encoded and used;
- non-deterministic control strategies can be implemented. Modules may not be executed in a predefined sequence, thus enabling the use of various reasoning paradigms;
- the system is transparent, *i.e.* it can provide natural explanations and justifications of its specific lines of thought and reasoning;
- it provides flexibility through incremental creation, debugging and updating of very large knowledge bases.

On the other hand, expert systems suffer from several short-comings which, in addition to the symbol grounding problem, include the so-called *completeness* issue. Using the example in Figure 2, a situation in which `Water_level > 10` and `Discharge = 55` is not represented in the knowledge tree. This would result in the failure of the inference engine to find an appropriate instance for `Action` and to draw the corresponding conclusions. Due to this problem, expert systems are said to be *brittle* systems, in the sense that as long as every question has an explicitly coded answer they will perform well, but as soon as a situation is not explicitly represented in the knowledge-base they fail quite suddenly, in a brittle fashion.

Applications of symbolic approaches can be exemplified by several recent publications such as Almeida and Schilling (1993) in which the construction of `IF...THEN...ELSE` rules in a knowledge base of a hydroinformatics system is described, while the most recent proceedings of the International Conference on Urban Drainage describe the application of expert systems to problems of urban hydrology quite extensively (see, for example, Ahmad *et al*, 1987; Delleur and Baffaut, 1990; Khelil *et al*, 1993; Martin-Garcia, 1995).

2.2 SOME SUB-SYMBOLIC APPROACHES

2.2.1 *Artificial neural networks*

Systems investigations, which Amorocho and Hart (1964) regarded as being concerned with the direct solution of technological problems subject only to the constraints imposed by the available data, and so not subject to 'physical' considerations, has recently undergone something of a renaissance, largely through the adaptation of artificial intelligence techniques, such as Artificial Neural Networks (ANNs) and Evolutionary Algorithms (EAs) (e.g. Babović and Minns, 1994). The particular advantage of the ANN is that, even if the 'exact' relationship between sets of input and output data is unknown - but is still acknowledged to exist - the network can be trained to learn that relationship.

The ability of the brain to perform difficult operations and to recognise complex patterns, even if these patterns are distorted by 'noise', has formed the subject matter of the discipline of cognitive psychology that has in turn strongly influenced the study of artificial intelligence (AI).The particular ability of the brain to learn from experience without a predefined knowledge of underlying physical relationships makes it an exceptionally flexible and powerful calculating device that AI researchers have long tried to mimic.

At the same time, other researchers have been devoted to reproducing, or modelling, physical phenomena by making use of electronic computational machines to solve ever-

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increasingly complex partial differential equations and related empirical relationships. These researchers are supported by a rapid increase in the computational capacity of modern computers and an emerging recognition of the advantages of massively parallel computation (parallel distributed processing) that performs the required calculations with ever-increasing speed. However, although the design and construction of the hardware for parallel computation is relatively straightforward, the software required for creating algorithms to utilise this parallel architecture for solving partial differential and other such equations efficiently is still quite limited.

These two groups of researchers, pursuing what appear to be quite different goals. have found a common ground in the field of artificial neural networks. One of the major applications of ANNs is in pattern recognition and classification or, more generally, system identification. In brief, an ANN consists of layers of processing units (representing biological neurons - *see* Hopfield, 1994) where each processing unit in each layer is connected to all processing units in the adjacent layers (representing biological synapses and dendrites). Many publications describe in much greater detail the architecture of various types of ANNs (for example, Beale and Jackson, 1990; Aleksander and Morton, 1990; Hertz *et al*, 1991). Figure 4 shows a schematisation of a typical multi-layer, feed-forward ANN.

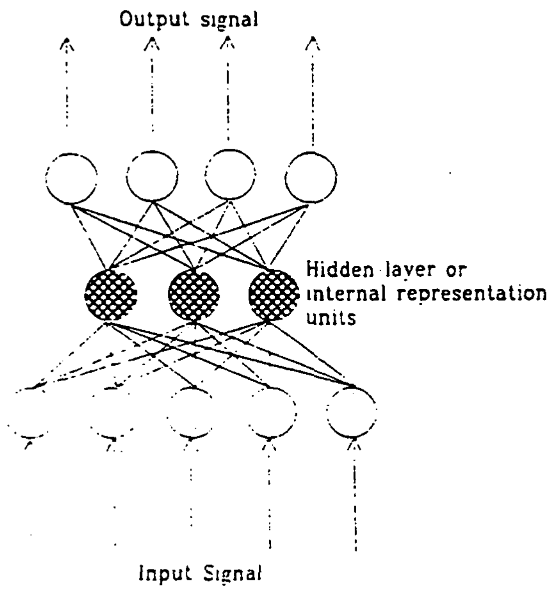

Figure 4 Representation of a multi-layer, feed-forward artificial neural network (ANN)

The working of an ANN can best be described by following the operations involved during training and computation. An input signal, consisting of an array of numbers x, is introduced to the input layer of processing units or nodes. The signals are carried along connections to each of the nodes in the adjacent layer and can be amplified or inhibited through weights associated with each connection. The nodes in the adjacent layer act as summation devices for the incoming (weighted) signals (Figure 5). The incoming signal is transformed into an output signal O_j within the processing units by passing it through a threshold function. A common threshold function for the ANN is the sigmoid function that is depicted in Figure 4 and defined as:

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$$f(x) = \frac{1}{1 + e^{-x}} \tag{1}$$

which provides an output in the range $0 < f(x) < 1$.

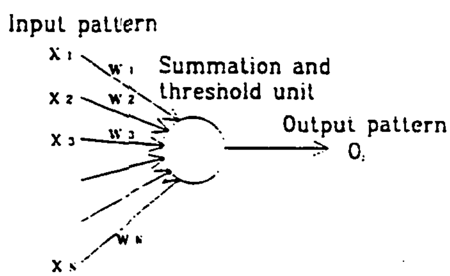

Figure 5 A typical ANN node

The output from the processing unit is then:

$$O_j = \frac{1}{1 + e^{-\sum x_i w_i}} \quad , \; 0 < O_j < 1, \quad \forall j \tag{2}$$

This output signal is subsequently carried along the weighted connections to the following layer of nodes and the process is repeated until the signal reaches the output layer. The one or more layers of processing units located between the input and output layers have no direct connections to the outside world and are referred to as *hidden* layers. The output signal can then be interpreted as the *response* of the ANN to the given input *stimulus*.

The ANN can be *trained* to produce known or desired output responses for given input stimuli. The ANN is first initialised by assigning random numbers to the interconnection weights. An input signal is then introduced to the input layer and the resulting output signal is compared to the desired output signal. An error or 'energy' function is then computed that represents the amount by which these two signals differ. This error function is defined as:

$$E = \tfrac{1}{2}\sum (D_j - O_j)^2 \tag{3}$$

where O_j is the network output and D_j is the desired output. The interconnection weights are then adjusted to minimise the error. This process is repeated many times with many different input/output tuples until a sufficient accuracy for all data sets has been obtained. A learning rule, known as the *generalised delta rule*, adjusts the weights associated with each connection by an amount proportional to the strength of the signal in the connection and the total measure of the error (*see* Rumelhart and McClelland, 1986). The total error at the output layer is then reduced by redistributing this error value *backwards* through the hidden layers until the input layer is reached. For this reason, this method is referred to as *error back-propagation*. The next input/output tuple is then applied and the connection weights readjusted to minimise this new

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error. This procedure is repeated until all training data sets have been applied. The whole process is then repeated again, starting from the first data set once more and continuing until the total error for all data sets is sufficiently small and subsequent adjustments to the weights are inconsequential. The generalised delta rule provides in fact a form of *gradient descent* method. where the energy function (3) is calculated and changes are made in the steepest downward direction. Although this method does not guarantee convergence to an optimal solution since local minima may exist, it appears in practice that the back-propagation method leads to solutions in almost every case (Rumelhart *et al*, 1994). In fact, standard multi-layer. feed-forward networks. with only one hidden layer have been found capable of approximating any measurable function to any desired degree of accuracy (Hornik *et al*, 1989). Errors in representation would appear to arise only from having insufficient hidden units or the relationships themselves being insufficiently deterministic.

2.2.2 *Evolutionary algorithms*

Evolutionary algorithms (EAs) are simulation engines of grossly simplified processes occurring in nature and implemented in artificial media such as computers. The fundamental idea is indeed the one of plagiarising natural processes. Darwinian theory of evolution depicts the adaptation of species to its environment as one of natural selection (Darwin. 1859). Perceived in this way. all species currently inhabiting our planet (and for that matter, all species that have ever lived on this planet) are actually the result of this process of adaptation.

Evolutionary algorithms in effect depict an alternative approach to problem solving. in which solutions to the problem are evolved rather than problems being solved directly. The family of evolutionary algorithms may be characterised by four main streams: Evolution Strategies (Schwefel, 1981), Evolutionary Programming (Fogel *et al*, 1966), Genetic Algorithms (Holland. 1975) and Genetic Programming (Koza, 1992).

Although different and applied for different purposes, all EAs share a common conceptual base. In principle, an initial population of individuals is created in a computer and allowed to evolve using the principles of *inheritance* (so that offspring resemble parents). *variability* (the process of offspring creation is not perfect - some mutations occur) and *selection* (more fit, or 'better', individuals are allowed to reproduce more often and less fit individuals less often so that the 'genealogical' trees of the latter will 'die out' with time).

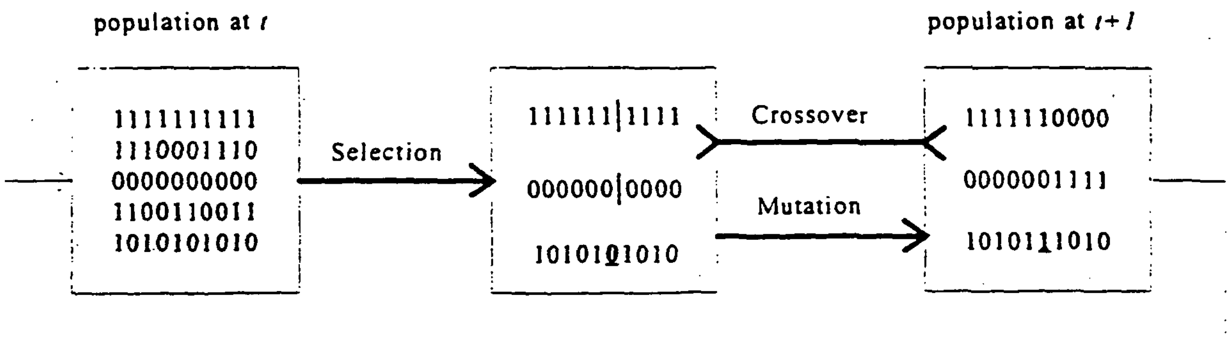

Figure 6 Schematic illustration of an evolutionary algorithm.

Figure 6 depicts the main processes that make up an evolutionary algorithm. From an initial. typically randomly generated, population of individuals the fittest entities are selected to be altered by genetic operators exemplified by *crossover* (corresponding to sexual reproduction) and *mutation*. Selection is performed on the basis of a certain fitness criterion in which the fitter individuals are selected more often. Crossover combines two genotypes by exchanging sub-strings around a randomly selected point. Mutation simply flips a randomly selected bit.

Similar to the processes of nature, one should distinguish between the evolving entity's genotype and its phenotype. The genotype is essentially a code to be executed (such as a code in the DNA strand in humans), and the phenotype represents the result of the execution of this code (such as a living person). The information exchange between evolving entities (parents) occurs at the level of the genotypes; however, it is the phenotypes in which we are really interested.

The phenotype is in effect an interpretation of a genotype in a problem domain. This interpretation can take the form of any feasible mapping. One of the main advantages of EAs is their domain independence. EAs can evolve almost anything, given an appropriate representation of the evolving structures. For example, for optimisation and constraint satisfaction purposes. genotypes are typically interpreted as independent variables of a function to be optimised. Several applications of genetic algorithms (GAs) that make use of this kind of mapping and with specific emphasis on water resources are described by Babović (1993).

In so-called *learning classifier systems (LCS)*, as introduced by Holland (1986), phenotypes take the appearance of rules in evolving knowledge-bases. LCSs are actually built on the top of ordinary GAs, and continuously augmented the knowledge-base with new and better-performing rules, thus avoiding a rigid and static tree structure. LCSs thus open avenues towards automatic model enhancement through the process of machine learning (*see* Wilson, 1994).

In genetic programming (GP), the evolutionary force is directed towards the creation of models that take a symbolic form. In this evolutionary paradigm, evolving entities are presented with a collection of data, and the evolutionary process is expected to result in a closed-form symbolic expression that describes the data. In principle, GP evolves tree structures representing symbolic expressions in Reverse Polish Notation. The nodes in this tree structure are user-defined. This means that they can be algebraic operators, such as *sin, log,* +, -, etc., or can take a form of *if-then-else* rules, making use of logical operators such as *OR, AND,* etc. (*see* Walker *et al*, 1993).

It is extremely difficult ,if not impossible, to describe the full potential of EAs and their applications in such a limited space. The reader is therefore referred to the original texts that describe the inner workings of EAs and their applications in much more detail. Here, however, we would like to highlight two essential properties of EAs:

- Evolutionary Algorithms are sub-symbolic models of computation. As was suggested before, the exchange of information between evolving entities occurs at the level of the genotypes. The phenotypes represent or contain the *meaning* encoded in the genotypes. This meaning (or semantic interpretation) is acquired through both a *mapping function* (from genotype to phenotype) and an interaction of the phenotypes with their environment. This applies for the entire EA family. GP in its most rudimentary form can be understood as a method for evolving trees which acquire meaning only when they are confronted with the problem domain;

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- The most important phenomenon in relation to EA performance is that it attains its knowledge about its environment through interaction with this environment. The knowledge about a problem that is being solved does not explicitly exist within the EA-based problem-solver before the problem-solving (*i.e.* evolutionary) process is initiated. This knowledge is acquired through the process of survival of the fittest. The consequence of this is that the process of solving problems actually transforms to one of adequately describing the problem and then letting the solution to the problem evolve itself.

3 Some Applications

3.1 RAINFALL-RUNOFF MODELLING

For rainfall-runoff modelling it is supposed that, subject to given antecedent conditions, there is an explicit relationship between the depth of rain falling on a catchment and the magnitude of the streamflow emerging from that catchment. Hall and Minns (1993) confirmed that for simple laboratory catchments and small sewered areas, an ANN is capable of learning the relationship between rainfall and runoff to a very high degree of accuracy even in the very simple case of having inputs restricted to antecedent rainfall depths and antecedent flow ordinates Figure 7 shows the results of an ANN model that was trained on actual data from a small urban catchment in the UK. The results of the ANN model have also been compared to the results of the single, conceptual, non-linear reservoir model called RORB.

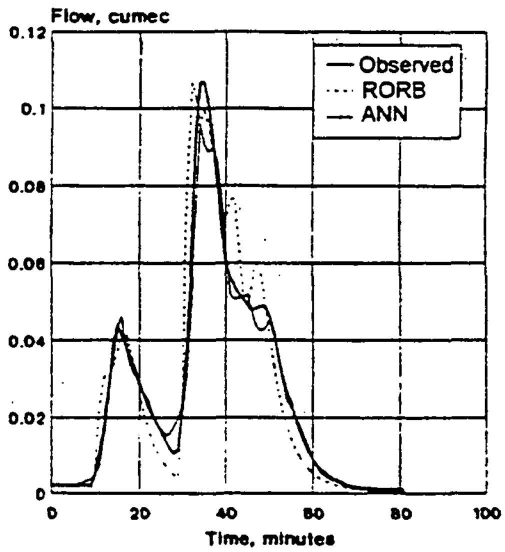

Figure 7 Comparison of ANN with conceptual model RORB

Minns and Hall (1995) continued these investigations into applications on more complex theoretical catchments exhibiting a range of behaviour patterns varying from the linear to the highly (in hydrological terms) non-linear and having inputs restricted to antecedent rainfall depths and antecedent flow ordinates (*see* Figures 8 and 9). The ANN model provides these exceptional results unhindered by constraints of volume continuity in the input and output data and, in fact, the units of the data are chosen simply for convenience of measurement and representation. Furthermore, simple, non-hydrological parameters like the percentage impervious area, may be easily incorporated into the model at the discretion of the modeller. These types of parameters may be derived from simple measurements or may even be highly intuitive, and are likewise unrestricted in terms of conditions of dimension or hydrological-physical consistency (*see* Minns, 1996).

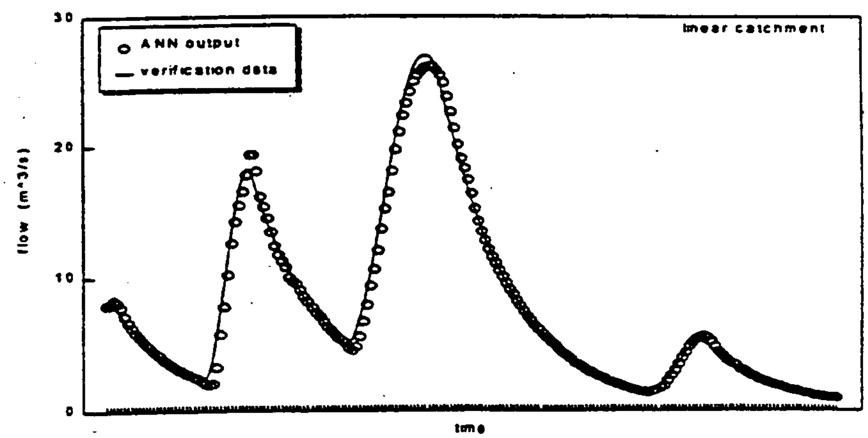

Figure 8 Verification of a 3-layer ANN for a linear catchment

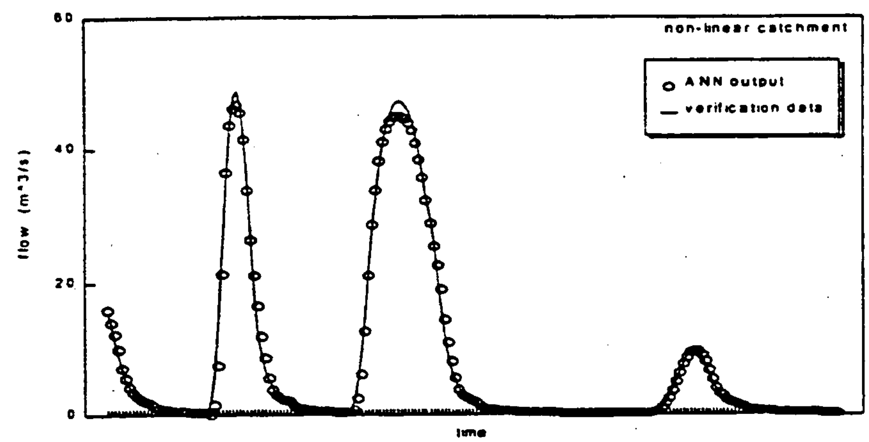

Figure 9 Verification of a 3-layer ANN for a non-linear catchment

Babović (1995, pp. 208-216) applied genetic programming techniques to induce symbolic expressions from the data used by Minns and Hall (1995). The best-performing expression for the linear catchment model (Figure 8) was:

$$q[t] = q[t-1] + 0.3r[t-1] - 0.29\sqrt{r[t-14]} \tag{4}$$

and for the non-linear catchment model:

$$q[t] = q[t-1] + r[t-2] - r[t-8] \tag{5}$$

where :

$q[t]$ denotes runoff in m^3/s at time t
$r[t]$ denotes rainfall in mm/hr at time t
t denotes time in hours.

Eqs. (4) and (5) performed with a similar accuracy to the ANN models of the same data. Although no attempt was made to interpret these equations physically, it is immediately obvious that the relevant variables emerging from the GP induced expressions might have something to do with the lag-time of the catchment. Extracting a semantic content from the ANN models is by no means as obvious however.

The potential role of ANN and GP models in hydrological modelling in general is manyfold. At the simplest level they may function as a flexible, easy-to-implement, lumped-conceptual models that relate rainfall data to runoff data for *individual catchments*. At the other end of the spectrum, they may be used to generate important components of physically-based, distributed hydrological modelling systems, whereby a sub-model of *individual physical processes* (e.g. unsaturated zone flow dynamics) is induced based only upon measured data. Such a sub-model may then replace whole systems of complex, non-linear, differential equations that would otherwise require great skills from the modeller to calibrate and powerful computing devices to solve.

3.2 MODEL CALIBRATION AND SYSTEM OPTIMISATION

With traditional conceptual hydrological modelling techniques the modeller applies his or her measured data together with some hydrological insight in order to adjust modelling parameters and equations manually and so eventually to calibrate the model. Babović *et al* (1994) describe the application of a genetic algorithm to the problem of model calibration. in which the genotypes of the GA are interpreted as roughness coefficients in a free surface pipe flow simulation and the evolution is directed towards the minimisation of discrepancies of model output and measured water level and discharge values - thus resulting in an automatic calibration of the roughness coefficients in the hydrodynamic model.

Solomatine (1995) explains that the process of calibrating a hydrological model is, in fact, a form of optimisation problem in which the objective function to be minimised is the difference (error) between computed output variables and the corresponding values measured in the physical system. The independent variables in the optimisation problem are the unknown model parameters. He further compares the performance of a GA to that of more traditional optimisation techniques and confirms the power of this methodology in global optimisation.

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Rapid selection of the optimal control strategy in a multi-objective water resources system is of primary importance for the real-time control of these systems. Masood-Ul-Hassan and Wilson (1995) describe how both ANNs and GAs can be used to improve system performance and respond to processes in the real world in the face of real-time constraints. They explain how an ANN can be trained off-line to replicate optimised gate-settings in a flood-control scheme. The generation of the optimised gate-settings with which to train the ANN was carried out using a numerical optimiser employing a traditional gradient descent method. This generation of the optimal control strategy data and the training of the ANN with this data is quite time consuming; however, after training, the ANN can be used instantaneously to recall the optimal gate settings corresponding to any given system state for which it was trained This paper further describes the implementation of a learning classifier system to the real-time control of a sewerage network. In a classifier system, actions in response to a given system state are obtained from a rule-based system. The learning classifier system improves its performance with time by generating new rules based on experience. A GA is used to generate the new rules through recombination of the best-performing classifiers that replace the low-performance classifiers.

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TOWARDS A DISTRIBUTED PHYSICALLY BASED MODEL DESCRIPTION OF THE URBAN AQUATIC ENVIRONMENT

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KEYWORDS: Urban drainage, sewer, groundwater, infiltration, geohydrology, modelling.

INTRODUCTION

Human use of land always affect the hydrological balance, and the use of land for urban development has pronounced effects on the local hydrology. The urban storm drainage and sewer systems interact with the local groundwater in a complicated manner. In many cases most of the rain falling over houses and gardens is drained by foundation drains and leakage to the sewers, often with substantially increased inflow/infiltration after a rainstorm as a result (Bäckman, 1985). Proper measures against these problems demand a geohydrological understanding of the draining process of the catchment studied.

Conceptual models are today widely used, and with success, for the modelling of these processes and their effect on the sewerage system. The conceptual model MOUSE NAM (DHI, 1994) has in Sweden been applied in more than 50 wastewater treatment plant catchments to model hydrological processes affecting the infiltration/inflow components in sewer networks (Gustafsson, 1995). However, one of the disadvantages with these conceptual models, is their incapability to fully consider existing overall knowledge of the catchment. Neither do they explain the physical underlaying reasons for certain results. This means that the effects from future measures in the urban nature, ie alternative drainage schemes etc, only to a limited extent, if at all, can be described with a conceptual model. But still, to analyse the effects of alternative measures is often one of the main reasons for applying models. Modelling tools, capable to describe the geohydrological processes in a more physical and distributed manner, are therefore needed to obtain more knowledge and understanding for these processes.

On the market today, physically based geohydrological model concepts already exists. One of these more advanced models is the MIKE SHE model (Abbot, 1986, and DHI, 1993), developed at the Danish Hydraulic Institute. MIKE SHE has up to now been widely applied on traditional surface and groundwater problems in water resources areas. In particular, MIKE SHE has shown to be a very capable tool when the effect of human

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interference is to be assessed. To examine to what extent MIKE SHE is practically applicable in urban modelling areas, a research project was carried out with support from the Swedish Water and Waste Water Works association. The overall goal was to test if it is possible to describe the surrounding geohydrological processes and their interaction with the sewer network, similar to the way dynamic pipe flow modelling can give a detailed description of the hydraulics.

VITTSKÖVLE - A SMALL VILLAGE WITH LARGE DRAINAGE PROBLEMS

The MIKE SHE pilot project was carried out in Vittskövle, a village outside the city of Kristianstad, Sweden. Vittskövle has about 250 inhabitants, mainly covered by residential district. The village has its own simple treatment plant (TP) with mechanical and biological treatment (figure 1). During the winter, some parts of the treatment process have to be stopped because of the very high inflows caused by a large amount of groundwater infiltration into the sewer network from foundation drains, leaking service pipes and main pipes. The average inflow to the TP is 5 l/s, with daily extreams exceeding 20 l/s during wet periods. These figures should be compared with a water consumption (foul flow) of 0.5 l/s, ie an average extent of dilution of 10.

During the mid 80´s, extensive pipe relining work was carried out, covering about 70 % of the main sewers (figure 1). The effect on the total infiltration was very low. Instead, locally risen groundwater levels in some cases led to local flooding problems during wet periods in areas with no foundation drains. The old leaking sewer network had before served the area as an efficient groundwater drainage system. The only traditional method that today seems to be the solution, is to build a complete new waste water system for the village, keeping the old system for drainage of the area. An estimate of the total cost for this solution is nearly US$ 10,000 per property, in total approximately US$ 1,000,000. These problems were among the reasons for looking at other more untraditional methods, ie new drainage schemes, reducing the natural groundwater flow and lowering the groundwater level in the urban areas. The question was where to build the new drains, and what the total effect on the infiltration would be.

MODELLING METHODOLOGY - MIKE SHE

MIKE SHE is a deterministic, distributed and physically based modelling system for simulation of hydrological processes in the land phase of the hydrological cycle. The model is applicable to a wide range of water resources and environmental problems related to surface water and groundwater systems and the dynamic interaction between these. The modelling package comprises a number of pre- and postprocessors to facilitate the input of data and the analysis of simulation results (DHI, 1993), among others: space interpolation routines; graphical editing; and plots of the variations in space and time of any variable, as well as animation tools. MIKE SHE simulates the variations in hydraulic heads, flows and water storage on the ground surface, in rivers and in the unsaturated and saturated subsurface zones. The areal variation of meteorological input data and catchment characteristics are represented in a network of grid squares. Within every grid square the soil profile is divided into a series of vertical layers.

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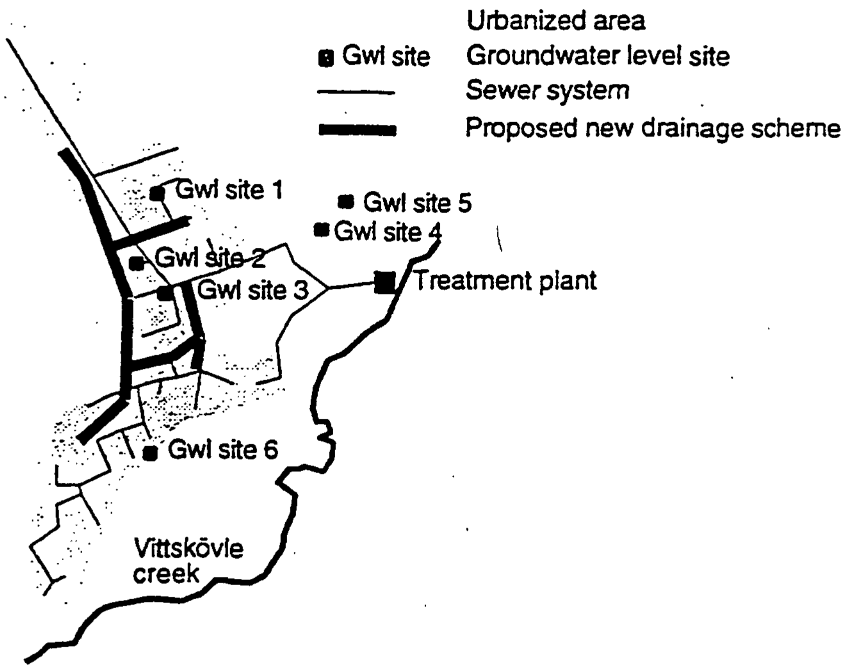

Figure 1. Principle sketch of Vittskövle village, including sewer system, TP location, and groundwater level observation sites. In addition, the location of the most efficient drainage scheme, found from simulations, is shown (see below).

THE PILOT PROJECT EXECUTION - SETTING UP AND VERIFYING THE MODEL

The catchment covers approximately 2 km^2. A horizontal discretization of 20 by 20 m was chosen, giving 60 by 85 horizontal grid nodes. The topography was described with discrete elevations collected from maps. In addition, the surface characteristics were described by parameters for roughness etc, as well as distribution codes for landuse and crops. The creek flowing through the catchment was described with input of cross-sections, bottom levels, roughness, as well as river bed lining charachteristics. The upper soil characteristics (ie porosity and hydraulic conductivity) were described for each type of soil profile. Four vertical simulation layers were used to describe the saturated zone, mainly corresponding to the geological layers, ie an upper sand layer (5 m thick) including silt and clay lences, a lower sand layer (20 m thick), a limestone rock layer (150 m thick), and an underlying glauconite sand (150 m thick) with artesian ground water. The geohydrological conditions for each of these layers were described.

Finally, the drainage function of the sewer network was described as grid squares with different drainage levels corresponding to pipe bottom levels, and different drainage coefficients describing the ability of sewer infiltration. With these two parameters, the amount of infiltration to the sewer system from each grid square is calculated with the following simple equation: $Q = A \cdot (GW - L) \cdot K$, ie by multiplying the grid square area (A) with the drainage coefficient (K) and the difference between groundwater level (GW) and drainage level (L) for the grid square. This description is certainly a conceptualization of the very complex physical behavoiur in the soil volume represented by the

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lumped grid square of 20 by 20 m. However, as long as reliable physical information about how cracked pipes and manholes are, is not available, which it very seldom is, a more physical approach on this micro scale is not applicable. But, the conceptualization of course encounter a calibration procedure to obtain reliable drainage coefficients.

The main input time series used in the simulations are precipitation and temperature. In addition, to describe the evaporative demand, the normal yearly variation of potential evaporation, leaf area index and root depth for different landuse were used. These input time series, used for both verification and other simulations, covered in total 15 years, ie 1979 to 1993. The model was verified by using, in addition to input time series, long term time series of observed inflow to the TP and observed ground water levels at six different sites inside the catchment (figure 1). While the sewer drainage process is based on a conceptual approach, the drainage coefficients has to be seen as empirical model parameters that has to be calibrated by a comparison between simulated and corresponding observed flows. The sum of the sewer drainage flow was calibrated against the long term time series of observed TP inflow. In addition, short term distributed observations of flow at different sites in the sewer network were used in order to obtain a reliable distribution of the drainage coefficients in different sub-areas of the sewer network. Short term observations had been carried out during 1980, ie before the extensive pipe relining work. An additional extensive measurement campaign, covering more than 20 observation sites, was carried out during March 1995, a period with very high groundwater levels. This was an important activity, which served the model with an up-to-date distributed description of the amount of infiltration in the different sub-areas after the pipe relining work. For the later stage (see below), when the most optimal locations of new drainage schemes were to be assessed, this information was endeed crucial.

Figure 2 shows the verification results for the inflow to the plant. It should be noticed, that comparable verification results easily could have been obtained by using a conceptual one-cell model (eg MOUSE NAM), maybe even better. However, the effects from future changes in the system could not have been described by that kind of model, which was the objectives of this study. Figure 3 shows the corresponding verification results for the groundwater tables variation in time and space in the area.

SIMULATED EFFECTS FROM HISTORICAL AND ALTERNATIVE MEASURES

Different simulation cases were studied in order to evaluate the effects from both historical measures and alternative future alleviation schemes. The results were quite interesting. If the pipe relining work would have been fully successful, giving a fully watertight system in areas relined, including service pipes etc, a reduction of the inflow to the plant by 45 % would have been possible. The real outcome of the relining work was a reduction of the inflow to the plant with approximately 15 %. However, it was in fact fortunate that the outcome of this work was partly unsuccessful, because the simulations of a fully watertight system, or natural conditions (no sewer network), shows that the exploitation made, would not have been possible without drainage of the catchment. Without drainage, small lakes would appear in the central lower parts during wet winter periods. Older maps of the area confirms this since parts of the outer skirts of the existing village are marked as marsh land. At the same time, simulated groundwater depths for present conditions shows expressively how the sewer network keeps the groundwater table at a low level along the sewers, ie along the properties.

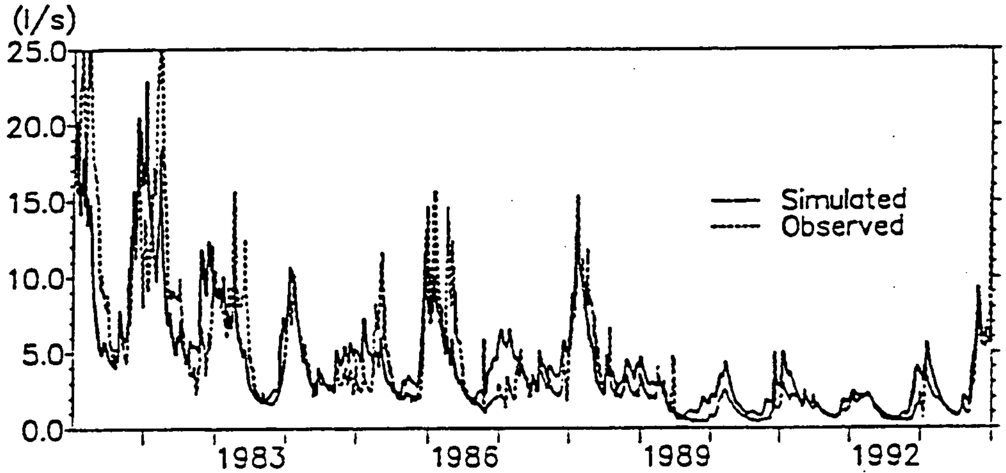

Figure 2. Comparison between simulated drainage flow and observed TP inflow.

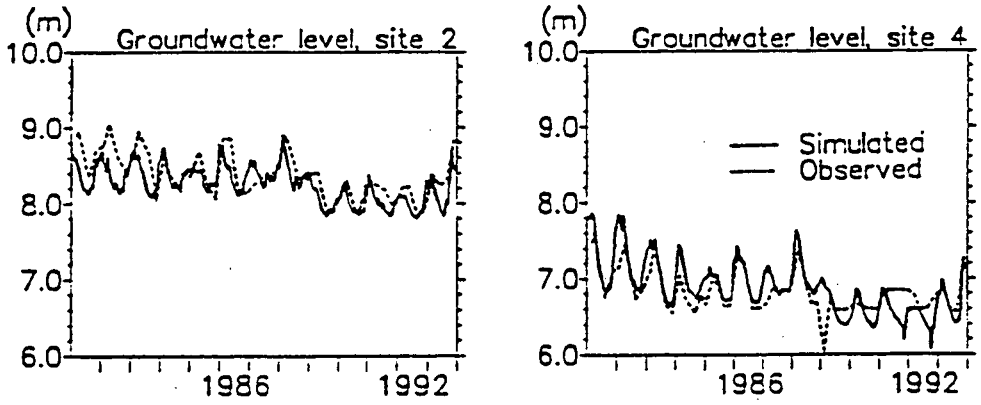

Figure 3. Comparison between simulated and observed groundwater levels at two sites.

Based on the main groundwater flow directions for the present system, and the detailed information concerning major sewer infiltration areas, a large amount of potential new drainage schemes were tested. The evaluation criterias included TP inflow, risk of waste water spill from sewers during periods with low groundwater levels (reverse infiltration), and risk of drying out the unsaturated zone (root zone). The most optimal drainage scheme found from the simulations is shown in figure 1, and covers a length of 1400 m with a depth of 1.5 to 3.0 m. The simulated effects with these drains indicate a reduction of the inflow to the plant by as much as 75 %, ie from the present average inflow on

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5 l/s to 1.2 l/s, without risk of increased groundwater levels. The cost estimates of these new drains, mainly located outside the urbanized areas, do not exceed US$ 100,000, which represents only a fraction of the costs for constructing a new sewerage system.

CONCLUSIONS

The application of MIKE SHE in Vittskövle shows expressively the possibilities and applicability of the model in urban areas. Compared with more simple conceptual models, MIKE SHE provides the possibility to analyse the effects from future changes in the geohydrological system. However, a distributed physically based model like MIKE SHE also demand much higher geographical resolution of the input, increasing the time for setting up the model, but of course, consequently, permitting higher geographical resolution of the output. Also, a distributed physically based model can be seen as a set of conceptual models communicating with each other. In the final end it is up to the user to define the number and size of these conceptual elements, permitting a simplyfied description of the natural system, with fewer and larger elements, as well as a more complex description where the conceptual elements only appear in micro scale.

Although MIKE SHE is fully capable of describing the required phenomena, some modifications are desirable for the specific problems in urban areas. Among the more important are: capability to describe the sewer network as a river with "closed" cross-sections and specific drainage coefficients for each pipe; better description of impervious areas and their connection to specific nodes in the sewer network; and more flexible connection of the drainage flow from a grid square to specific sewer network nodes. Based on a combination of further development of MIKE SHE and integration with other more specific tools dedicated for urban drainage, there is certainly a potential for MIKE SHE to act as the frame for the tomorrow urban drainage modelling system, encapsulating the complete urban aquatic environment.

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Recent developments of the Système Hydrologique Européen (SHE) towards the MIKE SHE

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Abstract The development of the Système Hydrologique Européen (SHE) started in 1977 as a joint effort by three European organizations: Institute of Hydrology (UK), the French consulting firm SOGREAH and the Danish Hydraulic Institute. The SHE is often quoted in the literature as a prototype of the distributed, physically based group of models. In this paper the comprehensive results of further developments of the MIKE SHE version, which have taken place during the last five years, are summarized. MIKE SHE simulates water flow, water quality and soil erosion processes for the entire land phase of the hydrological cycle. It is intended for scientific and engineering hydrology. MIKE SHE is a fourth generation, user-friendly modelling package comprising a number of comprehensive pre- and post-processors including digitizing, graphical editing, contouring, grid-averaging and graphical presentation with options for display of animations.

INTRODUCTION

The European Hydrological System, SHE, was developed as a joint effort by the Institute of Hydrology (UK), SOGREAH (France) and the Danish Hydraulic Institute (DHI). It is a deterministic, distributed and physically-based modelling system for describing the major flow processes of the entire land phase of the hydrological cycle. A description of SHE is given in Abbott *et al.* (1986a; 1986b). Further presentations of the joint experience and methodology of model application are given in Refsgaard *et al.* (1992), Jain *et al.* (1992) and Lohani *et al.* (1993). Since 1987 SHE has been further developed independently by three organizations which are the University of Newcastle (UK), the Laboratoire d'Hydraulique de France (LHF) and the DHI. The University of Newcastle's version, denoted SHETRANS, has been further developed and is presently used in Newcastle for research purposes. DHI's version of SHE, known as MIKE SHE, represents significant new developments with respect to user interface, computational efficiency and process descriptions. LHF has made an agreement with DHI on marketing and application of MIKE SHE in France and a few other countries.

MIKE SHE

Integrated modular structure

The core of MIKE SHE is the module that describes the water movement in the area

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under consideration – MIKE SHE WM. A number of add-on modules are available and can be applied according to the specific problems in the study area.

The following add-on modules are already available, or are under development, for water quality, soil erosion and irrigation studies:
(a) MIKE SHE AD – advection and dispersion of solutes;
(b) MIKE SHE GC – geochemical processes;
(c) MIKE SHE CN – crop growth and nitrogen processes in the root zone;
(d) MIKE SHE SE – soil erosion;
(e) MIKE SHE DP – dual porosity; and
(f) MIKE SHE IR – irrigation.

Below, a brief introduction to the WM module is given. For a further up-to-date description of the various modules, references are made to DHI (1993a), DHI (1995) and VKI (1995).

Water movement module (MIKE SHE WM)

The overall model structure is illustrated in Fig 1. MIKE SHE WM comprises six process-oriented components, each describing the major physical processes in individual parts of the hydrological cycle and, in combination, describing the entire hydrological cycle:
(a) interception/evapotranspiration (*ET*);
(b) overland and channel flow (*OC*);

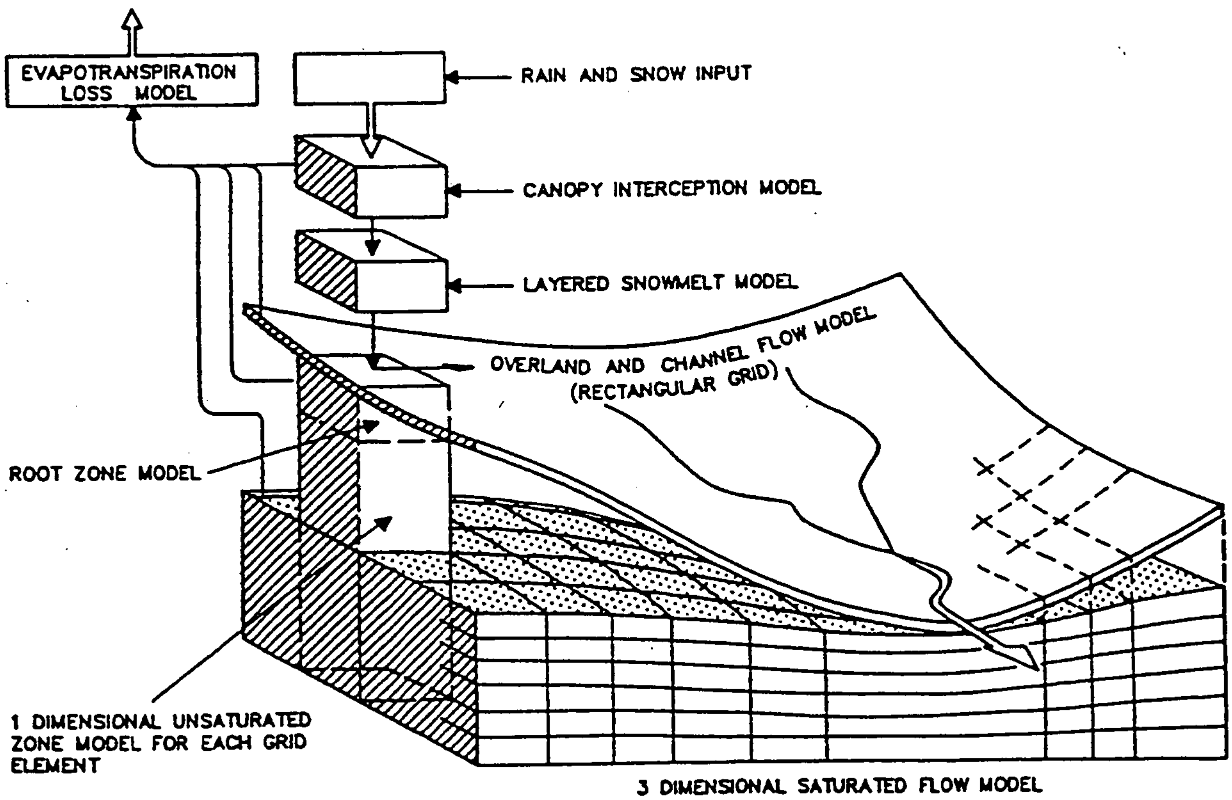

Fig. 1 Schematic representation of the components of MIKE SHE.

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(c) unsaturated zone (*UZ*);
(d) saturated zone (*SZ*);
(e) snowmelt (*SM*); and
(f) exchange between aquifer and rivers (*EX*).

The modular form of system structure, or architecture, ensures great flexibility in the description of the individual physical processes. Thus, some of the components already include alternative options for describing the processes. Data availability or specific hydrological conditions may favour one model description as compared to another. By ensuring that the data flow between components is unchanged, alternative methods which are generally accepted in a certain geographical region or a country, can be included in the MIKE SHE WM system, if required. The user can produce his own model configuration, e.g. starting with simple process descriptions and gradually changing to more complex descriptions, as and when required.

The governing partial differential equations for the flow processes are solved numerically by efficient and stable finite difference methods in separate process components. All process descriptions operate at time steps consistent with their own most appropriate temporal scales. Hence the processes may be simulated using different time steps which are automatically updated during the simulation and coupled with the adjoining processes as and when their time steps coincide. The facility allows for a very efficient operation, making it possible to carry out simulations for long time periods.

Individual components can also be operated separately to investigate a single process. This may be relevant in a range of applications, where only rough estimates of data exchange from other parts of the hydrological cycle are required. An example could be a groundwater study where only approximate recharge estimates may be required and a full coupling to the unsaturated zone above the groundwater table is unimportant.

The ability to provide an integrated description of the various processes, despite different time scales, is the most important feature of MIKE SHE WM. This integration has probably been the largest problem encountered during its development and provides a unique feature. Perhaps the most difficult coupling is the one between the unsaturated zone and the groundwater components, which is described in Storm (1991).

Pre- and post-processor

The user interface of the MIKE SHE module includes powerful pre- and post-processing facilities with particularly strong GIS capabilities. The software can be applied to the input data and results of all the MIKE SHE modules.

Some examples of the software capabilities are:

(a) digitization of contours from maps such as those of the ground surface and geological layers;
(b) digitization of the river system layout;
(c) digitization of aerially-distributed information, such as land use, soil types, etc.;
(d) transformation of vectorized information to grid information, e.g. interpolation of contour or point information;
(e) graphical editing of 2-D data and river data;
(f) very flexible transformation of geological/hydrogeological vectorized data into 2-D or 3-D grids;

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(g) double mass plots;
(h) water and solute mass balance calculations for any sub-area;
(i) arithmetic operations on matrices and time series;
(j) isoline plots, vector plots;
(k) plots of the variations in space of a variable in any layer or along any line through the model; and
(l) plots of time series of any variable.

All graphical presentations can be in colour and are produced with a UNIRAS graphical package.

The user can easily design, produce and display animation of any variable. This provides a unique opportunity to present the dynamic behaviour of the simulated system and adds a new dimension to error handling and interpretation of results.

APPLICATION EXPERIENCE

MIKE SHE is today being used operationally by a large number of organizations in different countries, ranging from university and research organizations to consulting engineering companies.

The original SHE version has been tested on a number of research catchments and applied to a few other projects, see e.g. Bathurst (1986), Storm *et al.* (1987), Refsgaard *et al.* (1991) and Refsgaard *et al.* (1992) for further details.

MIKE SHE is an extended version of SHE and has been applied to a large number of projects during the past few years. A list of applications in which DHI has been directly involved are shown in Tables 1 and 2 for research and consulting projects, respectively. These applications illustrate the very wide range of water resources problems for which MIKE SHE is a suitable tool.

In addition to the applications listed in the two tables MIKE SHE is used by other organizations in a large number of projects which are not known to the authors.

SHE was originally developed with a view to describing the entire land phase of the hydrological cycle in a given catchment with a level of detail sufficiently fine to be able to claim a physically-based concept (Abbott *et al.* 1986a; 1986b). The equations used in the model are, with few exceptions, non-empirical and well-known to represent the physical processes at the appropriate scales in the different parts of the hydrological cycle. The parameters in these equations can be obtained from measurements as long as they are compatible with the representative volumes for which the equations are derived.

In most regional catchment studies carried out so far, it has not been possible to represent the spatial variations in catchment characteristics with such a detail that the model could be considered physically-based. In fact, this was realized at an early stage when the applications changed from testing against analytical solutions and small-scale research areas to applications on medium sized catchment areas.

However, experience shows that the spatial resolutions and variations in properties used provide a very good representation of the conditions in the areas modelled. In practice the spatial variations are derived from maps describing topography, soil and land use patterns and interpreted geological conditions, combined with information about the general properties of the different map units. The model's parameter values are then modified during calibration to match observed conditions at discrete points.

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Table 1 List of MIKE SHE applications on externally funded research projects.

Location	Project Title	Period	Topics and references
Germany	The quantitative and qualitative impacts of real-time control on surface waters and storm water drainage and infiltration for water supply in Berlin Friedrichshagen	1994-1996	Groundwater pollution from river and sewer system. Demonstration of real-time control possibilities. Coupling with MIKE 11 river modelling system and MOUSE urban drainage modelling system
UK	Modelling of the impact of contaminated land on water quality using the MIKE SHE model	1994-1995	Groundwater pollution, geochemical modelling
Denmark	Strategic environmental research programme	1993-1996	Groundwater pollution
EU	Development of a European soil erosion model	1992-1994	Soil erosion. Styczen and Lørup (1994).
Sweden	Effects of forestry drainage and clear-cutting on flood conditions	1991-1992	Impacts of human activity on floods.
EU	Modelling of the nitrogen and pesticide transport and transformation on catchment scale	1991-1994	Agricultural pollution. Styczen and Sørensen (1994).
Denmark	Research programme on groundwater pollution from waste disposal site	1988-1990	Groundwater pollution. Bjerg *et al.* (1992), Brettmann and Jensen (1993), Jensen *et al.* (1993).
Denmark	Research programme on nitrogen, phosphorous and organic matter	1986-1990	Simulation of nitrogen on catchment scale. Coupling with daisy crop growth and nitrogen model. Hansen *et al.* (1991); Styczen & Storm (1993).
Denmark	Validation of pesticide models	1994-1996	Leaching of pesticides in clayey soils with preferential flow paths

There are a number of fundamental scale problems which need to be carefully considered in the model applications. This is particularly important when describing the interaction between the surface flow and the subsurface flows. A few areas where scale problems are encountered include:

(a) The interaction between groundwater and river. Since the flow is based on Darcy's law using the gradient between the river water level and the groundwater heads in the adjacent grid squares, the flow rates and the resultant head changes will depend on the spatial resolution used. This is an important aspect in, e.g. simulating the hydrograph recessions correctly.

(b) In catchments with a dense drainage network it is often not possible to represent the entire drainage system (many streams are of ephemeral nature). For such situations sub-grid variations in the topography need to be accounted for in order to simulate the hydrograph response in the main streams correctly.

(c) For modelling of infiltration and vertical unsaturated flow in the soil, the hydraulic parameters used in Richards' equation can be obtained from laboratory measurements on small undisturbed soil samples. However, for grid squares covering large areas (e.g. 25 ha) these are seldom representative unless completely homogeneous conditions exist in the horizontal directions. Therefore effective or representative parameters are used, which means that the simulated soil moisture conditions cannot be verified directly.

In fact much of the criticism against MIKE SHE often arises from the way the unsaturated flow is simulated and very seldom from how the groundwater conditions are treated.

For most catchment simulations, the use of Richards' equation becomes conceptual rather than physically based and simpler approaches could be chosen. Nevertheless, this equation provides a good routing description, and the capability to simulate capillary rise under shallow water table conditions is an important option for studies where, e.g. wetland areas are included. For situations where Darcy's law does not apply, a simple macro-pore option is included in the solution.

Because representative parameter values are used, the reliability of the results depends very much on the data available for comparison of the simulated spatial and temporal variations with observations. This is well-known from groundwater applications, where the aquifer properties (conductivities or transmissivities) are derived based on calibration against observed head variations in discrete points. For regional catchment studies, the model performance is usually evaluated based on comparisons against river discharges and groundwater heads. Very seldom are measured soil moisture data available, and if they are, such comparisons require that site specific properties are

Table 2 List of MIKE SHE applications on consultancy projects with DHI involvement.

Location	Project Title	Period	Topics
Estonia	Tapa Airbase - groundwater model	1993	Groundwater pollution
Slovakia	Danubian lowland – groundwater model	1992-1995	Surface water quality, river and reservoir erosion and sedimentation, groundwater quality and geochemistry (redox), wetland ecology coupling with ARC/INFO and Informix
Denmark	Six projects on optimization of remedial measures for safeguarding groundwater resources from pollution from waste disposal sites	1992-1994	Groundwater pollution
UK	River management study, River Avon, Wessex	1992-1993	Effects of groundwater abstraction and augmentation schemes on streamflow
Denmark	Environmental impact assessment of a highway construction	1992-1993	Effects of groundwater drawdown due to tunnel construction
Australia	Irrigation salinity	1991-1993	Process simulation of an irrigation district with focus on flow and salinity transport
Denmark	Two projects on identification of new well field for water supply	1991-1994	Groundwater, effects of abstraction on streamflows and wetlands
Hungary	Assessment of pollution hazards in groundwater supplies	1991	Groundwater pollution
Denmark	Water supply planning in Aarhus county	1988-1990	Groundwater resources, effects of abstractions on hydraulic heads and streamflows

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known.

It is often stated that distributed models require a large amount of data and are therefore very time consuming and complicated to set up and calibrate. In fact, a number of short-term screening evaluation projects have been carried out with MIKE SHE, e.g. in connection with studying the contamination risks from waste disposals. In these studies only sparse existing information about the hydrogeological conditions was available. The model was used to obtain an improved knowledge about the possible flow patterns around the waste disposal site based on the existing geological interpretations. These applications could also be used to identify where existing knowledge is lacking and assist in defining an appropriate monitoring programme.

Another common argument against distributed models is the risk of over-parameterization. This risk is, of course, always there. However, the general experience is that if the data to describe the spatial variations in the catchment are lacking, it is too time-consuming and not worthwhile to modify a large number of parameter values in order to improve, e.g. hydrograph predictions. In such cases very few parameters are used in the calibration and the reliability of the results are evaluated with this in mind.

ONGOING RESEARCH

From the above application records it appears that MIKE SHE has already been used comprehensively both for research studies and for practical routine applications. These applications reflect that for certain types of studies there is no adequate alternative to an integrated, distributed, physically based modelling approach like MIKE SHE. Nevertheless, it is realized that MIKE SHE, in its present form, is far from being complete as a tool for advanced hydrological analyses. Many problems, both practical and fundamental, need to be solved through future research activities. A very significant part of the research carried out recently in the international scientific community is, in fact, of direct relevance and most valuable in this context.

At DHI research and developments related to MIKE SHE is carried out in the following fields:

(a) Improvement of process descriptions. Research work on macro-pore flow and solute transport is presently being undertaken. Other activities such as inclusion of density effects in the groundwater component and description of hysteresis phenomena in the unsaturated zone are planned in the coming years.
(b) Improvements in numerical efficiency are taking place continuously.
(c) An interface to geographic information systems (ARC/INFO) is being developed.
(d) Coupling with DHI's generalized river modelling system MIKE 11 (DHI, 1994) is on-going. A coupled MIKE SHE/MIKE 11 enables description of sediment transport and water quality processes in the river system, as well as description of complex river and canal systems with hydraulic control structures. A typical area of application is irrigation command areas, where networks of both irrigation and drainage canals exist together with a large number of different hydraulic regulating structures.
(e) Fundamental research on the establishment of rigorous methodology on parameterization, calibration and validation is urgently needed. Some first, small steps have been taken, as described in Refsgaard *et al.* (1992), Jain *et al.* (1992) and DHI

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(1993b).

(f) Fundamental research on scale problems related to spatial variability of hydrological parameters is urgently needed. In particular, problems related to different scales used for data sampling, process description and model discretization need to be addressed. Although comprehensive international research is carried out in these years no operational results and conclusions are evident.

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Validation and applicability of distributed hydrological models

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Abstract In recent years, distributed, physically based hydrological models are being developed and applied more and more. At the same time, contradictions are emerging regarding claims of model applicability on the one hand and lack of validation of these claims on the other hand. In this connection the necessity of applying a rigorous validation methodology becomes obvious. The present paper presents an outline of a general methodology with special emphasis on the additional requirements for distributed models as compared to lumped rainfall-runoff models. A case study on validation and inter-comparison of three different models on catchments in Zimbabwe is described. The three models represent a lumped, conceptual model (NAM), a distributed, physically based model (MIKE SHE) and an intermediate approach (WATBAL). On the basis of these results and the authors' other experience, a discussion is given on the applicability and associated validation requirements for different types of hydrological models.

INTRODUCTION

In recent years, distributed, physically based hydrological models are being developed and applied more and more. At the same time contradictions are emerging regarding claims of model applicability on the one hand and lack of validation of these claims on the other hand. Hence, the credibility of the advanced models can often, with good reasons, be questioned.

Many different definitions of model validation are presently used. A consistent terminology with a set of definitions for terms such as conceptual model, computerized model, verification, validation, domain of applicability and range of accuracy is given by Schlesinger *et al.* (1979). Slightly different definitions are used by other authors, e.g. Konikow (1978), Tsang (1991) and Anderson & Woessner (1992).

Oreskes *et al.* (1994), using a different terminology to that of the other authors, state that verification and validation of numerical models of natural systems is theoretically impossible, because natural systems are never closed and because model results are always non-unique. Instead, models can be confirmed.

To a very large extent the contradictory statements among the above mentioned authors can be referred to differences in terminology. Furthermore, they illustrate the fundamental lack of a rigorous methodology for model validation.

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OUTLINE OF VALIDATION METHODOLOGY

Modelling protocol and definitions

A procedure describing the sequence of steps in a hydrological model application is often denoted as a modelling protocol. The protocol illustrated in Fig. 1 is a slightly modified version of the one proposed by Anderson & Woessner (1992). The key terms shown in Fig. 1, for which no unique, widely accepted definition exists, are here defined as follows:

(a) A *modelling system (= code)* is a generalized software package, which can be used

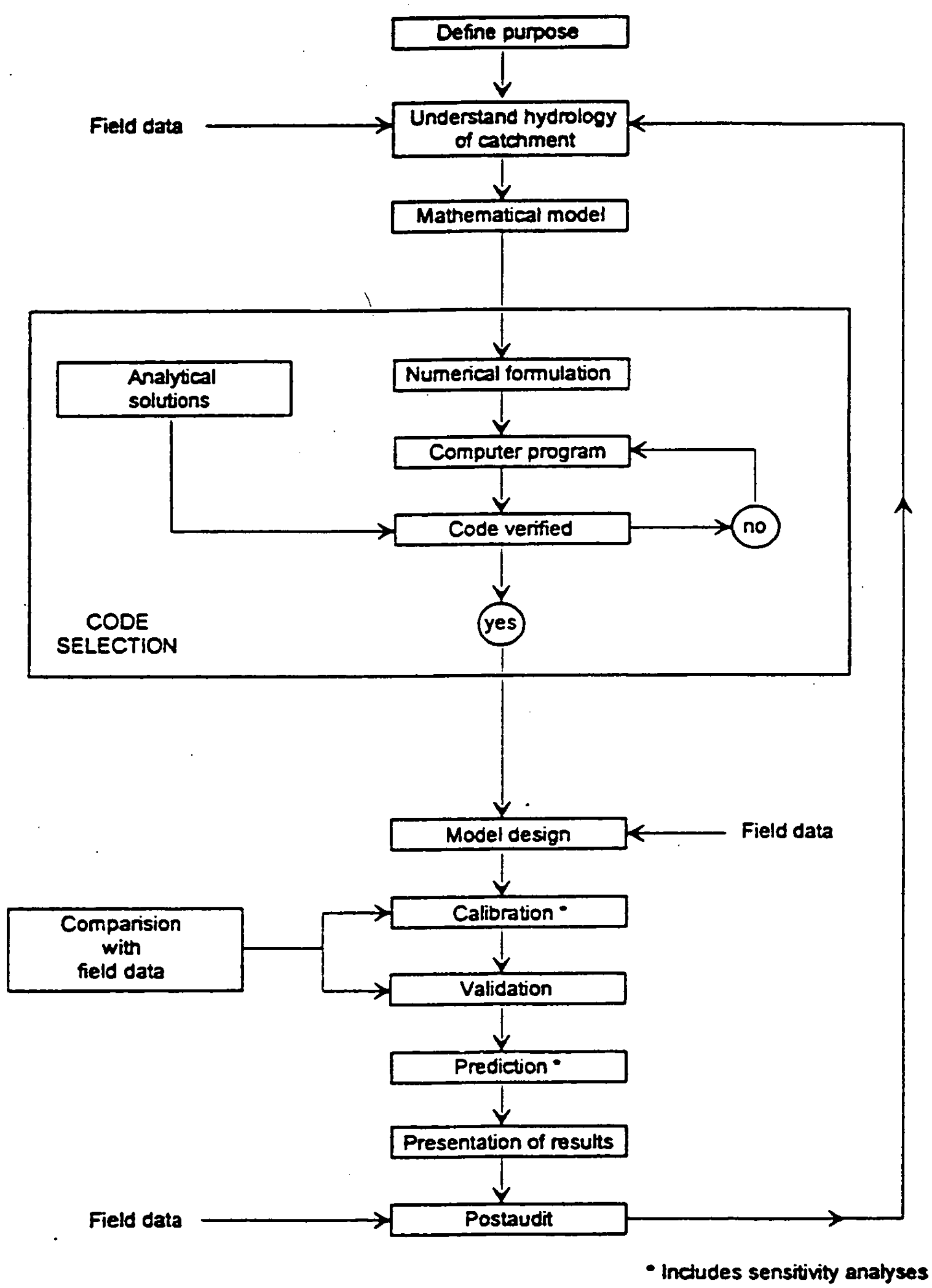

Fig. 1 The different steps in hydrological model application – a modelling protocol.

for different catchments without modifying the source code. Examples of modelling systems are MIKE SHE, Sacramento and MODFLOW.

(b) A *model* is a site-specific application of a modelling system, including given input data and specific parameter values. An example of a model is a hydrological model for the Elbe catchment.

(c) A modelling system or a code can be *verified*. A code verification involves comparison of the numerical solution generated by the code with one or more analytical solutions or with other numerical solutions. Verification ensures that the computer program accurately solves the equations that constitute the mathematical model.

(d) Model *validation* is here defined as the process of demonstrating that a given site-specific model is capable of making accurate predictions for periods outside a calibration period. A model is said to be validated if its accuracy and predictive capability in the validation period have been proven to lie within acceptable limits or errors. It is important to note that the term model validation refers to a site specific validation of a model. This must not be confused with the more general validation of a generalized modelling system, which will never be possible.

Testing scheme for validation of hydrological models

The testing of hydrological models through validation of independent data has, for a long time, been emphasized by the World Meteorological Organization (WMO). In their pioneering studies (WMO, 1975, 1986) several hydrological modelling systems were tested on the same data from different catchments. The actual testing, however, only included the standard split-sample test comprising an initial calibration of a model and subsequent validation based on data from an independent period.

The hierarchial testing scheme of Klemes (1986) appears suitable for testing the capability of a model to predict the hydrological effect of climate change, land use change and other non-stationary conditions. Klemes (1986) distinguished between simulations conducted for the same station (catchment) used for calibration, and simulations conducted for ungauged catchments. He also distinguished between cases where climate and land use are stationary and cases where they are not.

This combines to the definition of four basic categories of typical modelling tests:

(a) *Split-sample test:* calibration of a model based on 3-5 years of data, and validation on another period of similar length.

(b) *Differential split-sample test:* calibration of the model based on data before catchment change occurred, adjustment of the model parameters to characterize the change and validation of the subsequent period.

(c) *Proxy-basin test:* no direct calibration allowed but advantage may be taken of information from other gauged catchments. Hence, validation will comprise identification of a gauged catchment deemed to be of similar nature as the validation catchment, initial calibration, transfer of the model including adjustment of parameters to reflect actual conditions within the validation catchment, and validation.

(d) *Proxy-basin differential split-sample test:* Again no direct calibration is allowed but information from other catchments may be used. Hence, validation will comprise initial calibration of the other relevant catchment(s), transfer of the model to the validation catchment, selection of two parameter sets to represent the period before and after the change, and subsequent validations of both periods.

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EXAMPLE OF MODEL VALIDATIONS FOR CATCHMENTS IN ZIMBABWE

The following is based on results from a research project conducted at the Danish Hydraulic Institute (DHI, 1993a), where three different modelling systems have been validated and compared at catchments in Zimbabwe.

Three modelling systems

The following three modelling systems have been used for the Zimbabwe study:
(a) NAM: a lumped, conceptual rainfall-runoff modelling system (Nielsen & Hansen, 1973);
(b) WATBAL: a semi-distributed hydrological modelling system (Knudsen *et al.*, 1986); and
(c) MIKE SHE: a fully distributed, physically based modelling system (Abbott *et al.*, 1986a, 1986b; DHI, 1993b).

NAM and MIKE SHE can be characterized as very typical of their respective classes, while WATBAL falls between these two classes. All three systems are being used on a routine basis at the Danish Hydraulic Institute (DHI) in connection with consultancy and research projects.

Selected catchment

The three catchments in Zimbabwe which have been selected for the model tests are Ngezi-South, Lundi and Ngezi-North. A brief data collection/field reconnaissance to Zimbabwe was arranged to obtain relevant information. This included basic hydro-meteorological data, comprising daily series of rainfall and runoff, and monthly series of pan evaporation. Detailed information on land use was obtained through sub-contracting the University of Zimbabwe to prepare land use maps based upon 1:25 000 aerial photographs, while information on soil and vegetation characteristics was collected from various institutions and relevant literature. Furthermore, 1:50 000 topographical maps were collected and digitized. Finally, available data on vegetation characteristics, soil characteristics, hydrogeology and water rights were obtained. A more detailed description is given in DHI (1993a).

Testing scheme

The testing scheme for the Zimbabwe study involved all four types of tests proposed by Klemes (1986). In the present paper, results from the following two tests are shown:
(a) Split-sample test based on data from Ngezi-South, comprising an initial calibration of the models and a subsequent validation using data for an independent period.
(c) Proxy-basin test including transfer of the models to the Lundi catchment, adjustment of parameters to reflect the prevailing catchment characteristics and validation without any calibration.

The testing of the three models has been undertaken in parallel.

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Performance criteria

For measuring the performance of the models for each test a standard set of criteria has been defined. The criteria have been designed with the sole purpose of measuring how closely the simulated series of daily flows agree with the measured series. Due to the generalized nature of the defined model validations, it has been necessary to introduce several criteria for measuring the performance with regard to water balance, low flows, peak flows, and so forth.

The standard set of performance criteria comprises a combination of graphical plots and numerical measures. The graphical diagrams used include joint plots of the simulated and observed hydrographs, scatter diagram of monthly runoffs, flow duration curves and scatter diagram of annual maximum discharges. To support the graphical presentations, various numerical measures are computed, including the overall water balance, the Nash-Sutcliffe coefficient (*R2*) and an index (*EI*) measuring the agreement between the simulated and observed flow duration curves. Furthermore, additional measures for each hydrological year are computed.

The coefficient *R2*, introduced by Nash & Sutcliffe (1970), is computed on the basis of the sequence of observed and simulated monthly flows over the whole testing period (perfect agreement for $R2 = 1$):

$$R2 = 1 - \sum_{m=1}^{M} \left(Q_m^o - Q_m^s\right)^2 / \sum_{m=1}^{M} \left(Q_m^o - \bar{Q}^o\right)^2 \qquad (1)$$

where M = total number of months; Q_m^s = simulated monthly flows; Q_m^o = observed monthly flows; and $\bar{Q}^o$ = average observed monthly flows over the whole period. The flow duration curve error index, *EI*, provides a numerical measure of the difference between the flow duration curves of simulated and observed daily flows respectively (perfect agreement for $EI = 1$):

$$EI = 1 - \int \left| f_o(q) - f_s(q) \right| \mathrm{d}q \Big/ \int f_o(q)\,\mathrm{d}q \qquad (2)$$

where: $f_o(q)$ = flow duration curve based on observed daily flows; and $f_s(q)$ = flow duration curve based on simulated daily flows.

The above criteria thus measure the extent to which the models are able to provide an accurate representation of the overall water balance. Furthermore, they characterize the overall accuracy of the simulated series of monthly flows and its capability to represent the overall pattern of daily flows. In spite of its incompleteness, the above criteria provide a reasonable summary of the overall model performance.

Results of model validations

(a) Split-sample test: This test is based on data from Ngezi-South and comprises an initial calibration of the models and a subsequent validation using data for an independent period. The main results are summarized in Table 1.

As indicated by this table, the performance of all three models is generally very similar. All models are able to provide a close fit to the recorded flows for the calibration period, while for the independent validation period, the performance is somewhat reduced, as expected. The reduction is, however, limited, and all models are able to

Table 1 Ngezi-South: summary of split-sample test results (all flows in mm year^{-1}).

Calibration			$Q_{SIM} - Q_{OBS}$		
Year	Rain	Q_{OBS}	NAM	WATBAL	MIKE SHE
1971-1972	890	131	−19	−11	−11
1972-1973	317	2	1	−1	7
1973-1974	1290	349	14	−5	−10
1974-1975	1087	236	5	8	2
Mean:	896	179	0	−2	−3
Mean $\lvert Q_{SIM} - Q_{OBS} \rvert$:			10	6	8
R2			0.97	0.96	0.97
E1			0.88	0.95	0.91
Validation			$Q_{SIM} - Q_{OBS}$		
Year	Rain	Q_{OBS}	NAM	WATBAL	MIKE SHE
1975-1976	879	90	3	24	5
1976-1977	872	116	−28	2	7
1977-1978	1131	245	26	37	67
1978-1979	609	59	−13	2	−9
Mean:	873	128	−3	16	18
Mean $\lvert Q_{SIM} - Q_{OBS} \rvert$:			18	16	22
R2			0.89	0.86	0.84
E1			0.74	0.86	0.80

maintain a very good representation of the overall water balance, the inter-annual and seasonal variations, as well as the general flow pattern.

(b) Proxy-basin test (ungauged catchment): This test comprises a transfer of models to the Lundi catchment, adjustment of parameters to reflect the prevailing catchment characteristics, and validation without any calibration.

The proxy-basin test was arranged to test the capability of the different models to represent runoff from an ungauged catchment area, and hence no calibration was allowed prior to the simulation. For all models three alternative runoff simulations were prepared, reflecting an expected low, central and high estimate, respectively.

All models used the experience from the Ngezi-South calibrations, in combination with the available information on the particular catchment characteristics for Lundi. While the NAM model used this information in a purely subjective manner to revise model parameters, both the WATBAL and MIKE SHE models used this information directly for the model setup. The estimates prepared by the latter two models have, however, also been influenced by the individual modellers' subjective interpretation of the available information on soil and vegetation characteristics.

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As an example of the model performance, the hydrograph simulated by MIKE SHE is shown in Fig. 2 for two of the years, together with the flow duration curves and the scatter diagram of monthly discharges for the entire five year validation period. A summary of the main results of the proxy-basin tests is given in Table 2.

In general, all models provide an excellent representation of the general flow pattern, yet with some discrepancies for the small and/or larger peaks. As seen in Table 2, the best runs of the individual models provide a good representation of the overall

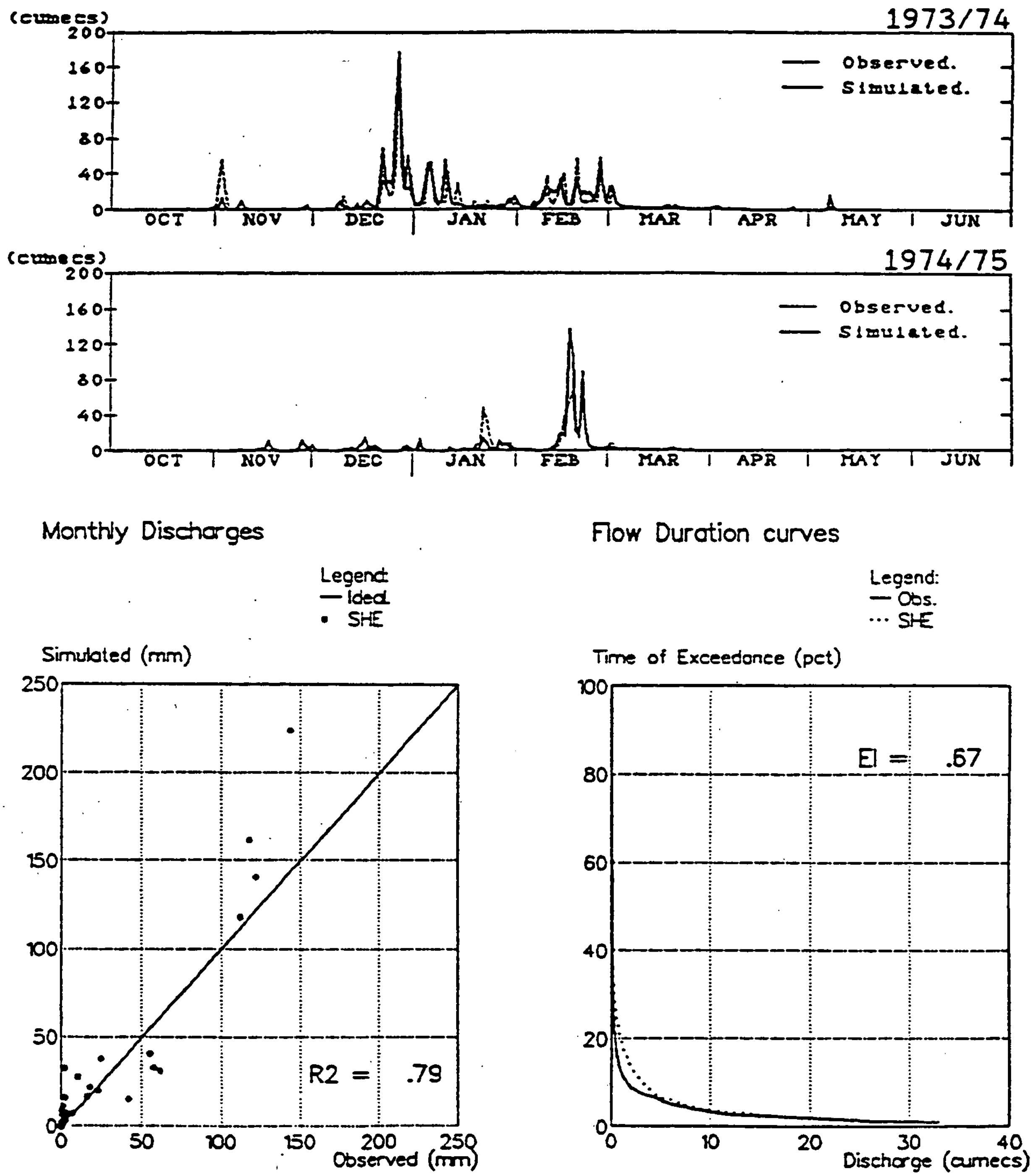

Fig. 2 Lundi MIKE SHE (central estimate) proxy-basin test hydrographs from two of the years together with the flow duration curve and scatter diagram of monthly discharges for the entire period.

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Table 2 Lundi: summary of proxy-basin test results (all flows in mm year^{-1}).

Calibration			$Q_{SIM} - Q_{OBS}$								
Year	Rain	Q_{OBS}	NAM			WATBAL			MIKE SHE		
			Low	Centre	High	Low	Centre	High	Low	Centre	High
1971-1972	920	89	5	63	122	1	3	32	−7	27	29
1972-1973	371	2	2	5	10	−1	−1	6	10	21	21
1973-1974	1384	460	−44	72	151	−78	3	34	43	98	120
1974-1975	1046	217	−36	44	116	−55	−17	14	8	45	61
1975-1976	857	89	−23	23	77	−14	−11	11	−20	13	22
Mean:	915	171	−19	42	95	−29	−5	19	7	41	51
Mean $\mid Q_{SIM} - Q_{OBS} \mid$:			22	41	95	30	7	19	18	41	51
R2			0.89	0.87	0.57	0.85	0.91	0.90	0.86	0.79	0.71
E1			0.66	0.72	0.41	0.59	0.76	0.75	0.78	0.67	0.63

water balance, while maintaining the significant inter-annual variability to a satisfactory degree and an overall good simulation of flows within individual months.

The overall performance of the central estimates by the NAM and MIKE SHE models is somewhat reduced, compared to validation runs for the Ngezi-South catchment, as expected when no calibration is possible. The estimates would, however, still be very valuable for all practical purposes. For the WATBAL model, the central estimate is even better than that obtained for the validation period for Ngezi-South, providing a very accurate representative record of observed runoff.

Conclusions on the model validations on the Zimbabwe catchments

The following conclusions are based primarily on the results shown above; however some of the other results of the original research project (DHI, 1993a) are also referred to.

For the split-sample test, the NAM, WATBAL and MIKE SHE models generally exhibit similar performance. All models are able to provide a close fit to the recorded flows for the calibration period, without severely reducing the performance during the independent validation period. Hence, this test suggests that if an adequate runoff period for a few (3-5) years exists, any of the models could be used as a reliable tool for filling in gaps in such records, or used to extend runoff series based on the long-term rainfall series. Considering the data requirements and efforts involved in the setup of the different models, however, a simple model of the NAM type should generally be selected for such tasks.

For the proxy-basin tests, designed for validating the capability of the models to represent flow series of ungauged catchments, it had been expected that the physically-based models would produce better results than the simple type models. The results, however, do not provide unambiguous support for this hypothesis. All three models generated good results, with the WATBAL providing slightly more accurate results than the others. Hence, for the Zimbabwean conditions, the additional capabilities of MIKE SHE, namely the distributed, physically-based features relating to subsurface flow, proved to be of little value in simulating the water balance.

For the proxy-basin tests it is noticed that the uncertainty range represented by the low-high estimates is significantly larger for the NAM than for the WATBAL and MIKE SHE cases. This probably reflects the fact that parameter estimation for ungauged catchments is generally more uncertain for NAM, whose parameters are semi-empirical coefficients without direct links to catchment characteristics.

In summary, the results of the comprehensive validations suggest that, given a few years of runoff measurements, a lumped model of the NAM type would be a suitable tool from the joint point of view of technical and economic feasibility.

For ungauged catchments, however, where accurate simulations are critical for water resources decisions, a distributed model is expected to give better results than a lumped model, if appropriate information on catchment characteristics can be obtained.

APPLICABILITY AND DIFFERENT VALIDATION REQUIREMENTS FOR LUMPED AND DISTRIBUTED MODELS

In accordance with the above case study from Zimbabwe, it is our general experience

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that distributed models are generally not better tools than lumped models for rainfall-runoff simulation when they are only calibrated and validated on runoff data from the outlet of the catchment. Even in cases when calibration and validation is possible at some internal discharge stations, the runoff simulations cannot generally be expected to be significantly better than those from well calibrated lumped models.

Consequently, lumped and distributed models should be applied complementary to the lumped models used for runoff simulations in cases where calibration is possible, and distributed models should be used for other more difficult issues, such as simulation of groundwater/surface water interactions, effects of land use change and water quality.

Due to fundamental differences in model structure, modes of operation and objectives of application, the validation requirements are much more comprehensive for distributed models, as compared to lumped models. The differences are summarized in Table 3, from which the need for multi-criteria, multi-scale validation criteria appears.

Acknowledgements The modelling work on the Zimbabwe catchments were carried out by the following staff members from DHI: Jesper Knudsen, Børge Storm, Merete Styczen and Roar Jensen. During the data collection and field reconnaissance in Zimbabwe, kind help and assistance was provided by ministries, institutions and individuals.

Table 3 Illustration of need for incorporation of multi-criteria and multi-scale aspects in methodology for validation of distributed models.

Model type	Lumped, conceptual	Distributed, physically based
Output	At one point: * Runoff => *single variable*	At many points: * Runoff * Surface water level * Ground water head * Soil moisture => *multi-variable*
Success criteria (excluding problem of selecting which statistical criteria to use)	Measured <=> simulated * Runoff, one site => *single criteria*	Measured <=> simulated * Runoff, multi-sites * Water levels, multi-sites * GW heads, multi-sites * Soil moisture, multi-sites => *multi-criteria*
Typical model application	Rainfall-runoff * stationary conditions * calibration data exist	Rainfall-runoff, unsaturated zone, groundwater, basis for subsequent water quality modelling Impacts of man's activity * non-stationary conditions sometimes * calibration data do not always exist
Validation test	Usually "split-sample test" is sufficient => *well defined practise exists*	More advanced tests required: * Differential split sample test * Proxy basin test => *need for rigorous methodology*
Modelling scale	Model: catchment scale Field data: catchment scale => *single scale*	Model: depends on discretization Field data: many different scales => *multi-scale* problems

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Modeling the Effects of Management Practices on Nitrogen in Soils and Groundwater

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I. INTRODUCTION

During recent years, an increasing nitrate load has been registered in many countries in subsurface water (soil and groundwater), in surface waters (rivers, lakes), and even in the sea. Increased levels of nitrate are caused by both point and nonpoint sources. While point sources in many countries are subject to control through legislation, the nitrate load from nonpoint sources has proven more complicated to deal with.

The analysis of leaching rates and concentrations in soils is obscurred by large seasonal and yearly changes because of natural fluctuations in, e.g., meteorological conditions, as well as of man-made influences from the agricultural practices, e.g., fertilizer rates, cropping patterns, general management practices, drainage, and irrigation. Furthermore, measurements of nitrate concentrations in rivers, which are relatively easy to carry out and often form part of a national monitoring program, are influenced not only by leaching from agricultural fields, but are also a result of many activities taking place within the upstream catchment in terms of land use, groundwater abstractions, drainage of wetland areas, etc. The specific hydrological and geological conditions in the catchment play, of course, an important role on the timing and magnitude of the load. Due to these different effects and interactions, it is extremely difficult to distinguish between natural and man-made variations and

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trends in agricultural pollution, and to pinpoint particular management actions which may have caused the observed nitrate levels.

If the pollution reaches levels where the effects become visible, such as increased eutrophication in lakes, fish death in fjords and seas due to oxygen deficiency, or the public health is threatened by high nitrate concentrations in the public drinking-water system, political decisions are bound to be taken, specifying a reduction in the agricultural pollution. However, in order to ensure that proper decisions are taken regarding water quality, it is necessary to identify the feasible options and evaluate their economic and environmental consequences. For that purpose a quantification of the effects of proposed measures is required.

Mathematical modeling is the most powerful approach for analyzing the present situation (e.g., pinpointing sources) and testing alternative management strategies. It provides a way to account for the characteristics of the natural system in a systematic manner as well as the spatial and temporal variations in input data, and to analyze the effects of changes in individual input data.

In contrast to point-source pollution, agricultural pollution is a problem on a large spatial scale, which always introduces a number of uncertainties in the modeling exercise. The modeler therefore needs a sound understanding of the hydrological, microbiological, and geochemical processes in order to be able to conceptualize the area under study, to estimate model parameters, and to evaluate and test model predictions. It is important that the users of the models ensure that the outcome of the model simulations answer the posed questions with sufficient certainty, given the constraints in input data and the available process descriptions. That is, the user must consider the probability of correctness and robustness of the results through methods such as statistical analyses and sensitivity analyses. It is equally important to emphasize to what extent the results obtained are site specific or how/if they can be generalized. A number of these issues will be discussed below.

II. MODELING OF NONPOINT POLLUTION

A number of phases need to be considered in connection with model applications of nonpoint pollution problems. Table 1 presents the most important ones, which probably cover most cases.

Although nonpoint pollution is a regional problem, it can be studied on many spatial and temporal scales. The problem-definition phase should clarify the purpose of the modeling exercise and the required scale.

Research studies are usually concerned with detailed analyses of processes over small distances—e.g., a single soil profile, a field, or a small drainage catchment. These studies provide an excellent framework for quantitative

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Table 1 Phases to Be Considered in Connection with Model Applications

Definition of the problem and objective of study
Understanding of the hydrological system
Understanding of the physical and microbiological mechanisms
Conceptualization of the system
Selection of mathematical model(s)
Collection of data
Estimation of parameters
Definition of management practices
Calibration/testing/validation of model(s)
Predictions based on chosen scenarios

descriptions of processes, implementation of these in model codes, and testing of models based on a good characterization of the system. For that type of study a field measurement program is established, providing input data as well as values of the dependent variables (water content, hydraulic head, concentration), with sufficient spatial and temporal detail to allow proper testing/validation of models.

Research studies can be extended to cover small, well-defined catchments with a well-defined mass balance. However, for projects focusing on larger geographic areas, such as studying the impacts of management practices on nitrate concentrations in groundwater aquifers or streams, extrapolation based on small-scale studies becomes difficult because such studies are likely to represent only a limited selection of the characteristics (soil types, depth of unsaturated zone, vegetation, etc.) found in the larger area. Models covering larger areas may therefore provide a better basis for decision making with regard to management strategies or policies. On the other hand, the problem here is how adequately to characterize the area under study. As the scale increases, the information required for running the models cannot be derived directly (e.g., from measurements), and the results become only approximate due to the simplifications introduced and the neglected spatial variability of certain features. In addition, validation is difficult. Site-specific comparisons against observed data cannot be made because representative (or effective) parameter values rather than measured parameter values are used. It is therefore of paramount importance that model users recognize and report the limitations and uncertainties in the model predictions. Table 2 describes the links among scale, purpose of study, and type of model that are involved.

Understanding of the system under study includes a listing and prioritization of pathways and processes taking place, relating to both water and

Table 2 Links Among Scale, Purpose of Study, and Type of Model Recommended for a Particular Project (e.g., Nitrate Movement)

Level	Vertical scale	Horizontal scale	Purpose	Type of model
(1)	Soil (root zone)	Point	Research	Leaching model
(2)	Soil (root zone)	Field	Research	Leaching model
(3)	Soil/groundwater	Field-research catchment	Research	Catchment model
(4)	Root zone	Regional/ catchment	Management	Leaching model
(5)	Soil/groundwater	Regional/ catchment	Management	Catchment model

nitrate, as well as considerations of how these particular conditions may be conceptualized. Considering a catchment as the study unit (Figure 1), water may be added to the field as rain or as irrigation (drip, sprinkler, furrow, flooding). The water may infiltrate or run off along the surface. If the water infiltrates, it may move uniformly through the soil, or part of the water may follow preferential pathways (macropores of different types). At larger depths, the groundwater may either contribute significantly to the river discharge with baseflow, or the process may be shunted by tile drains. The groundwater system itself may need thorough description, because the presence of geological layers with different hydrological properties will affect the actual flow paths.

Nitrogen may be added to a field as mineral fertilizer, as different types of manure, as aerial deposition, or even as a pollutant in irrigation water. The time of application is known to affect leaching. Nitrogen may leave the field in the form of NH_3 through volatilization. Mineral nitrogen may be taken up by plants; it may be immobilized and later mineralized. Denitrification of nitrate may also take place. Nitrate may be stored temporarily in the root zone, or leached out of the root zone. Most of these processes are dependent on the temperature, and some depend on the C pools or the water/oxygen content.

Below the root zone, denitrification can be negligible in the aerated part, whereas nitrate reduction due to the presence of organic matter, pyrite, or Fe(ii)-silicates must be expected in the saturated part. Furthermore, in some environments, groundwater with a natural load of nitrate occur, making it difficult to identify sources and sinks.

Wetland areas situated in the transition zone between the terrestrial and the aquatic systems along rivers may show a particularly high denitrification capacity, which is important to take into account when assessing the nitrate

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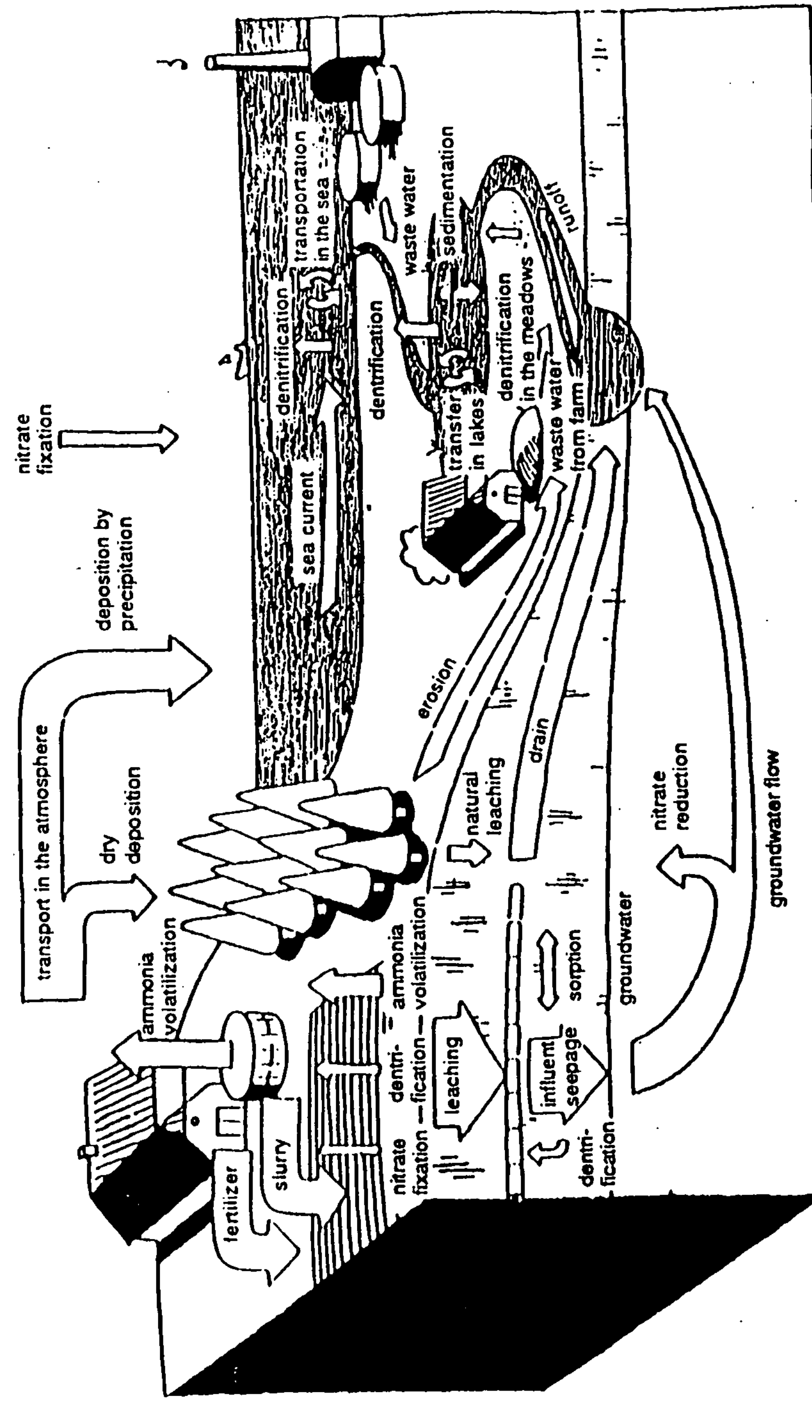

Figure 1 Transport and transformation of nitrogen and phosphorus in the environment. (*Source:* Ref. 28.)

load to the river system [1]. Some management plans utilize this and suggest development of wetlands in a narrow band along the river system.

It is important that proposed options for controlling leaching can be described by the model codes. As the economic consequences of such options are closely related to crop yields, proper evaluation of the options usually requires simulation of crop yields as well as simulation of soil N dynamics.

After the system has been conceptualized, appropriate model code(s), which can accommodate all the necessary processes, should be selected. For all types of modeling, it is important to define the correct boundary conditions. For example, leaching models usually have limited capabilities for dealing with surface runoff and ponding, as they seldom consider accumulation or runoff of water on the soil surface. The descriptions available for the lower boundary conditions are of particular importance when the groundwater table is found within the upper 1–2 m of the soil.

III. TYPES OF MODELS AVAILABLE

A large number of computer models have been developed in the past years to describe different parts of the hydrological system and different types of transport and reaction mechanisms. They vary in complexity, ranging from simple empirical formulas to fully distributed, physically based models. Traditionally, there has been a distinction between models describing the vadose zone and models describing the groundwater zone, and very few attempts have been made to integrate these model types to simulate the flow and transport from the source on the field to the recipient, e.g., the river or lake.

Vadose-zone models are usually one-dimensional leaching models describing the vertical flow and transport from the soil surface to the groundwater table. Groundwater models are usually two- or three-dimensional models representing the groundwater aquifer or a cross section of the aquifer. Although some groundwater models include a description of the interaction between the aquifer and rivers and other surface water bodies, usually they cannot describe the particular conditions which take place in wetland areas, such as the retention and removal of nitrate. This requires specific codes usually developed for ecological modeling.

It is outside the scope of this chapter to give a comprehensive review of existing models. The main emphasis here will be put on presenting a few state-of-the-art descriptions, which are believed to be the most promising tools not only for research studies but also for management problems.

A. Leaching Models

A large number of models describe the vadose zone, including the root zone, in a sir soil profile. The important leaching models are s lled mecha-

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nistic models, implying that they, as far as possible, describe the chemistry and physics of processes. Fourteen leaching models were reviewed by Willigen [2], and comparative tests of five models (ANIMO, RENLEM, EPIC, DAISY, SWATNIT) were carried out under the auspices of the EEC [3].

Examples of such models are SOIL-N [4], ANIMO [5,6], SWATNIT [7], and DAISY [8,9], which are all mechanistic and treat the majority of the water and nitrate processes described above. Although these models may appear to include the same or almost the same processes, there are differences which are important to emphasize. In some cases these differences may be of vital importance in determining which model is appropriate for a particular problem. Some differences among the models are summarized in Table 3.

SOIL-N and ANIMO both have to be used in conjunction with a hydrological model which can produce the necessary hydrological information (e.g., flow and moisture content). SWATNIT and DAISY include their own hydrological model, based on Richards' equation. All of the models describe the nitrate transport and transformation in two steps, by first carrying out the water flow calculations, then the N calculations. This approach may be acceptable where N is not a limiting factor, but in situations with serious N deficiency, it may pose problems. The plant growth simulated in the first step may be optimal from a water-availability point of view, but seriously restricted by N deficiency during the second run. The actual evapotranspiration calculated in step 1 will therefore be overestimated, resulting in wrong estimates of nitrate concentrations and fluxes. For both SWATNIT and DAISY, it is expected that new versions will carry out all calculations in one step.

Solute transport may be calculated as a product of flux and concentration (as in SOIL-N), or it may be calculated by means of the convection/dispersion equation (the other mentioned models). None of the four N-models include simulation of preferential flow paths at present, but some models exist which are able to consider mobile/immobile phases of water and solute. At least for new versions of SWATNIT and DAISY, inclusion of preferential flow is under consideration. It should be noted that preferential flow paths may have at least two different effects on N leaching: on one hand, a portion of the solute may move very quickly to considerable depth; on the other hand, solute already absorbed in the matrix may be bypassed by "clean" water, so leaching is delayed.

Heat transport is calculated in three of the models, while SOIL-N requires the results of a separate heat flow model.

All four models can accept different types of N fertilizer as input, and handle application of several doses of fertilizer over the simulation period.

The models describe the kinetics of mineralization as a first-order process, but the number of pools ranges from 2 to 7. ANIMO and DAISY take explicit accour of one or more pools of biomass. Nitrification is included in

Table 3 Comparison of Four Deterministic Models*

	ANIMO	SOIL-N	SWATNIT	DAISY
Two-step simulation	+	+	+(−)	+(−)
Solute transport	conv./ disp.	Flux • conc.	Conv./ disp.	Conv./ disp.
Pref. flow	−	−	−(+)	−(+)
Heat trp.	+	Separate module	+	+
Fertilizer type and dist. in soil	+	+	+	+
Mineralization	+	+	+	+
Nitrification	+	+	+	+
Denitrification	+	+	+	+
Ammonia		+		
volatilization	+	−	+	+
Oxygen status				
Expl. calculation	+			
Impl. calculation		+	+	+
No. of organic pools	4	2	2	7
Biomass pools	+	−	−	+
Growth module	Sketchy plant growth module	−	Elaborate plant growth module	Elaborate plant growth module
Plant uptake: trp. from bulk soil to root surface: vertical root distribution considered:	−	−	+	+
Considers uptake of 1. NO_3^- 2. $NO_3^- + NH_4^+$	2	1	2	2

*For explanation, refer to Section 3.1.

all of the models. So is denitrification, but different measures of oxygen status are used in the different models. Only ANIMO calculates the oxygen status of the soil air; SOIL-N and SWATNIT utilize a function of soil moisture content as an implicit measure of soil oxygen.

Adsorption and volatilization of ammonia are treated in ANIMO, SWATNIT, and DAISY, but fixation of ammonia in clay minerals is not included in any of the models.

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It is very important to simulate the plant uptake of nitrate correctly, as this is one of the major processes for removing nitrate from the soil profile. In some models (e.g., SOIL-N), the uptake is specified by the user, but uptake follows a logistic curve. This may be an advantage on research fields where this component can be assessed, but it hampers the use of the model for predictive purposes because the user must specify the expected future uptake of nitrate. The other models contain or may be combined with a growth module, where plant growth is limited by water and nitrate availability, and this enables the model to be used for predictions, provided no other nutrient limits the growth. The uptake may be calculated on the basis of transpiration fluxes (ANIMO), and assuming uniform concentrations at the root surface and in the bulk soil (SOIL-N, ANIMO). Another approach is to account for transport of water and solute from the bulk soil toward the root, using a steady-state approach (SWATNIT, DAISY).

The plant growth modules differ considerably among the models. ANIMO deals with plant processes only in a sketchy way, and, as mentioned, SOIL-N hardly includes them. SWATNIT and DAISY account for gross photosynthesis, respiration, distribution of dry matter and nitrogen over the different organs, etc.

The general conclusion of a test of 14 models [2] showed that prediction of nitrogen uptake by crop and dry-matter production requires one of the models with a detailed growth module. For both soil water and mineralization simulations, the results [2] showed that the detailed mechanistic models (represented by the four examples above) were not necessarily better than simpler models. They require detailed information about soil hydraulic and certain chemical properties, and are generally very sensitive to parameter values. On the other hand, they apply to a wide range of conditions. The simple models tested were generally applicable to a limited range of conditions only and cannot be used for extreme conditions.

The skill of modeling is thus to conceptualize the problem correctly, to assess the level of complication necessary, and to choose a model which allows for this [2,3]. On the other hand, a higher degree of reliability is not necessarily gained from choosing the most complicated model available.

B. Groundwater Models

Groundwater models describing the flow and transport mechanisms of aquifers have been developed since the 1970s and applied in numerous pollution studies. They have mainly described the advection and dispersion of conservative solutes. More recently, geochemical and biochemical reactions have been included to simulate the transport and fate of pollutants from point sources such as industrial and municipal waste-disposal sites.

Fewer attempts have been made to simulate nonpoint pollution of agricultural fertilizers and chemicals. The main problem arises from the fact that nonpoint pollution studies require characterization of physical, chemical, and biochemical characteristics over large areas. An additional problem in connection with groundwater modeling is to provide an estimation of the nitrate input to the groundwater through the unsaturated zone. If they are not based on results from leaching models, the timing and volume of nitrate fluxes are difficult to estimate because they depend on several factors, including the length of the unsaturated zone. This will have an important effect on the simulated concentrations in the groundwater. In areas with a shallow water table, the surface application pattern is reflected in the temporal variation in nitrate concentration of the groundwater to a higher degree than in areas with large distance to the groundwater table.

Models that include groundwater transport and chemical reactions cover a wide spectrum of codes with one-, two-, or three-dimensional presentation of the groundwater flow and including chemical reactions based on assumptions of equilibrium or kinetic degradation. Reviews can be found in [10–13]. Use of groundwater models for the study of nonpoint pollution problems is also described in [14] and [15].

Frind et al. (1990) [16] and Kinzelbach et al. (1991) [17] provide good examples of modeling studies of nitrate transport and nitrate reduction in the groundwater aquifer. Engesgaard and Jensen [18] and Storm et al. [19] made the first attempts to provide a fully three-dimensional simulation of the nitrate pollution in a groundwater aquifer. Their simulations were based on simplified chemical reaction calculations rather than using a kinetic reaction model.

There have been few attempts to couple the unsaturated and saturated zone processes. Bogardi et al. [20] presents a Dutch example where the ANIMO model was combined with a quasi-three-dimensional groundwater model, SIMGRO.

Storm et al. [19] and Styczen and Storm [21,22] coupled the DAISY model with the fully distributed catchment model MIKE SHE [23] to simulate the transport and transformation of nitrate in a catchment from the application on the fields to the river system through the unsaturated and saturated zones. Examples of results from this study is presented in Section VI.B.

C. Wetland Models

Modeling of the conditions in wetland areas in a relatively new discipline compared to the modeling of soil and groundwater systems [24]. One example of this type of model is given by Dørge [25], who developed a general model for freshwater wetlands to determine the retention and removal of nitrogen in this zone. The model consists of a simple hydrological submodel and a m-ˀ complex biological submodel included in a hydro- ·amic river model.

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IV. INPUT DATA FOR THE MODELS

A. Data Requirements

The amount of data needed to run comprehensive models as presented above tends to limit the use of such models. Because of the interdisciplinary nature of the models—covering the fields of agriculture, hydrology, geology, etc.—it is beyond any doubt that both data collection and model application requires input from many specialists with relevant complementary backgrounds. In particular, for parameter estimation it is important to emphasize how incomplete availability of input data will limit the reliability of the model results. Below is given a brief presentation of the most important types of data needed.

All models need the driving variables for the hydrological processes: precipitation, potential evapotranspiration, and air temperature, generally on daily basis. For growth models, global radiation is a common input.

With respect to soil hydraulic properties, a profile description is required, together with moisture characteristics from representative horizons, and values for hydraulic conductivity. As hydraulic conductivity always is a difficult parameter to measure and interpret (due to a very large variation within short distances), its value will often be subject to calibration in catchment applications. General soil data such as organic matter content, textural analysis, and volume weight are used directly or indirectly in most models.

For groundwater aquifers, the hydrogeological parameters, hydraulic conductivity and storage coefficients, are required for all important geological layers.

Measurements of the C content in the different horizons are usually the base for estimation of the size of (some of) the organic pools. The C/N relation of the organic matter is another important factor. The initial concentrations of nitrate and ammonia in the soil influence the simulation at the beginning of the run, but have little influence after 1 year. Wrong estimation of the organic pools, however, may influence the simulation results over many years. Turnover rates for different organic pools usually stem from laboratory work or long-duration field trials and are not changed from simulation to simulation.

Crop specifications differ from model to model, but it is necessary to know which crops are on the field during the simulation, the time of sowing, and the time of harvest. If a growth model is not included, specifications of parameters such as rooting depth and leaf area index are necessary to govern transpiration and plant uptake of nitrogen. In models with a growth module, the above variables are generated. In some models, crop yields are a necessary input.

With regard to management practices, the type, timing, and application method of fertilizers used on the field must be known. The C and N contents of mineral fertilizers are easy to assess, while for animal manure, slurry, etc., assessment is difficult. The C and N contents depend strongly on the storage method and time, as well as on influx of rain water to slurry tanks.

Tillage practices, particularly time of plowing, is important, as it influences the incorporation of organic material on the surface, as well as turnaround of the distribution of nitrate in the topsoil. Irrigation is another management practice which needs to be taken into account, with respect to type, amounts, and time of irrigation.

For the transport calculations, dispersion parameters are required to determine the spreading of the reactants.

The determination of the locations and extent of reaction zones in the groundwater aquifer is important for simulation of the total nitrate reduction. Information from well logs can provide information of the position of the redoxcline describing the boundary between the upper oxic zone containing O_2 and NO_3^- and a lower anoxic zone as illustrated in Figure 2. At this boundary, denitrification is likely to take place. If the reaction is fast, it may be appropriate to assume that all nitrate disappears when passing this boundary. For more complex chemical reactions, rate coefficients will be required.

B. Data for Model Calibration

The amount of data needed for model calibration depends on the scale of the problem. For leaching studies concerned with simulation of individual profiles, the models may be calibrated against measurements of soil tension and/or moisture content. Since it is often difficult to ensure correct simulation of water recharge, groundwater potentials and drainage flows from the area may be important information. Observations of solute concentrations in the soil will increase the reliability of the model but are seldom available except in research studies.

Catchment modeling requires information on groundwater heads and river flows for calibration of the water cycle. Observed NO_3^- concentrations in aquifer and river provide a basis for calibration and validation of the transport and reaction part of the model.

Comparisons between actual and simulated crop yields are a good indication of whether the growth simulation is reasonable.

Usually, some knowledge of the vertical concentration profiles in the groundwater can be used for calibrating the model with respect to dispersivities.

C. Data Availability

The data listed above are seldom available in totality, except on research plots. Here, nitrate concentrations at different depths or in drainage water may be available too, and may be used to check the simulations.

Monitoring programs are usually of a more extensive nature, and contain

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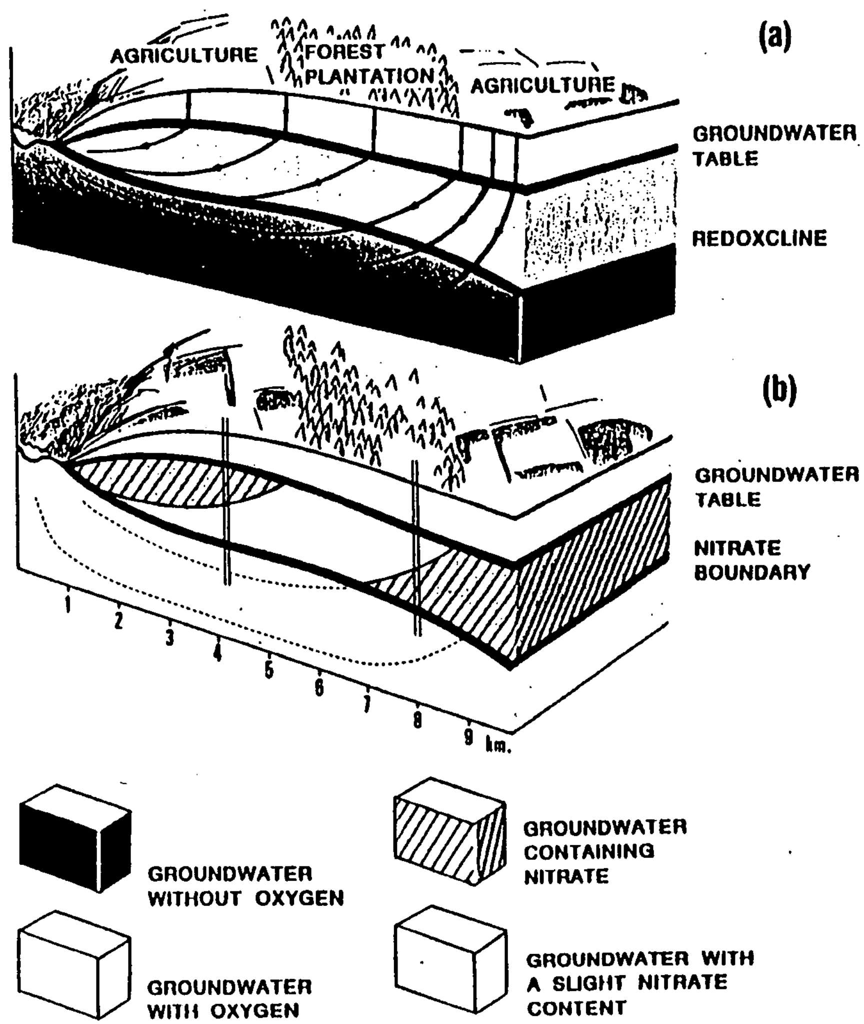

Figure 2 (a) Groundwater flow and potentials in a reservoir with unconfined groundwater. The redoxcline indicates the boundary between groundwater with and without oxygen. (b) Distribution of nitrate pollution below a sandy area with both agricultural and natural land uses. The nitrate leaching from the agricultural areas follows the stream lines to the redoxcline. (*Source:* Ref. 29.)

fewer measurements of variables such as soil moisture and tension. The areas covered by such programs are commonly larger, leading to less intensive knowledge concerning C and N contents of the soil, soil characteristics, content of C and N in manure spread in the area, etc. If, however, the monitoring program deals with N, usually the cropping pattern, the total manure application, as well as occurrence of nitrate in streams and drainage water are measured. Simulations of such areas require some qualified guesswork and assessments of which parameters can be generalized for larger areas.

For larger areas which are not covered by special programs, most data will have to be generated from statistics and general databases. Information may stem from sources such as meteorological records, soil maps, GIS databases, geological databases, statistical yearbooks, reports on fertilizer sales, local agricultural advisers, etc. Information which is not available locally (e.g., organic content of the soil, C/N relations) may be based on information from research plots in an area subject to similar cropping and fertilizer practices and generalized.

D. Parameter Estimation

The strategies for parameter estimation differ according to the choice of model (column/catchment), the scale of simulation, and the data availability.

Even for simulations of research plots of, e.g., 100 m^2, it must be realized that measured parameter values may not be precisely equal to the "effective" parameter which describes the plot best. Considerations concerning spatial variability are of relevance, as it must always be remembered that many measurements represent "a point" only.

The larger the area and the less specific the data, the greater is the need for specific upscaling strategies. For larger areas, the first step is generally to divide the area into uniform units with respect to soil types, vegetation, cropping pattern, etc., to systematize the parameter estimation. The next step is to decide to which extent representative data (mostly in the form of point measurements) are available for each of the defined units, whether "average" values may be used or spatial variability should be taken into account, and whether additional measurements must be initiated.

The modeler plays an important role as screener and selector of data, particularly for projects covering relative large areas that are not subject to a comprehensive measurement program. The model output is thus dependent not only on the model used, but also on the skills of the modeler with respect to choosing effective parameters and interpreting, e.g., statistical information for model use.

The estimated parameters may or may not be verified. If the flow in the unsaturated zone can be calibrated only against a groundwater level or drain-

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age flow, it is almost impossible to differentiate among values of hydraulic conductivity at different horizons in the profile. It can be argued that this is more important in small-scale studies than in large-scale studies, but it may nevertheless have an effect on both the timing of leaching and the amount of nitrate leached. This may lead to erroneous decisions concerning necessary management improvements.

Both retention curves and the hydraulic conductivity influences evapotranspiration and plant growth, and one of the ways to ensure that the values chosen are in the right range is to check the yields against measured yields. This is possible in models containing a growth module. Yield data are almost always available from research plots, and are recorded in statistical databases.

It is almost impossible to ensure the correctness of C and N contents used if they have not been measured. If the system is considered a stable system, the model may be run over a number of years, and the respective pools checked against the initial values. It is always a sound practice to start the simulation at least 1 year before the period in which one is interested, in order to get rid of initial errors. However, organic pools take a long time to reach a balance, and may influence the simulations for at least 5 years.

V. RELIABILITY OF RESULTS

The reliability of the results within a model depends on a large number of factors:

1. Inclusion of all the important processes in the model
2. Appropriateness of the process descriptions in the model
3. Uncertainty of parameter estimation
4. Interference from noncontrollable sources such as point sources e.g., silage tanks, manure heaps, septic tanks

Important processes which may not be included in a model could include macropore flow, drainage flow, or the lateral flow of a perched water table. The presence of macropore flow will tend to lead to an overestimation of the hydraulic conductivity of the profile, in order to simulate the runoff peaks observed in tile drains or the quick changes observed in the groundwater table. However, in reality the matrix conductivity may be quite small, and nitrate in the matrix may move much more slowly than simulated by the model.

Processes may be described at different levels of complexity. Too simple a description may cause errors. As an example, in simulations in clay soils with a high groundwater table, the use of a retention curve that consists only of a drying phase tends to overestimate the upward transport of water during a dry period (hy[illegible]esis).

Uncertainties due to parameter estimation are countless. A typical reason for uncertainty is the fact that it is difficult to take into account the variability of the area which is modeled. The variability may not be fully known, and even if it is known, it may be difficult and/or very time consuming to include the effects of variability of a range of parameters in a study. For some parameters, e.g., saturated hydraulic conductivity, point measurements may not represent the spatial average very well. However, measurements of solute concentrations in drainwater or groundwater represent an average for a certain area, and even column models are usually also taken to represent a given area (1 m–1 ha). Therefore the modeler has to search for parameters which represents an effective average for the area in question, taking into account the spatial variability.

VI. MODEL OUTPUT AND CASE STUDIES

This section presents two examples of model applications which in part have formed the basis for decision making. The first example shows the use of the leaching model DAISY to quantify differences among different cropping sequences under the same or different management practices. In the second example DAISY has been applied in combination with the MIKE SHE catchment model to simulate the transport and transformation on a medium-sized catchment using existing data.

A. Example One, Single-Column Simulation

A root-zone model may be used to highlight differences among different cropping sequences under the same or different management practices, or subject to different environmental conditions. Figure 3 [26] illustrates the effects of different fertilizer management systems. A cropping sequence containing winter wheat, sugar beet, barley with undersown grass, grass, and barley was repeated four times on two soil types (sand and sandy loam), using weather data from 20 years from western and eastern Denmark, respectively. This cropping sequence was subjected to five different fertilizer management practices. The amount of nitrogen available to the plants should be approximately the same in all treatments, but for treatment A, animal manure is applied during autumn; in B, all manure is applied during spring; for C1 and C2, manure application time varies from year to year (of the 5-year sequence, treatments includes three and two autumn applications, respectively); and in treatment D, only mineral fertilizer is applied. The management practices on the two soil types are identical except with regard to time of plowing. Spring plowing, which is practiced on the sand, is generally not practiced on the sandy loam due to the fact that these soils are too sticky during spring.

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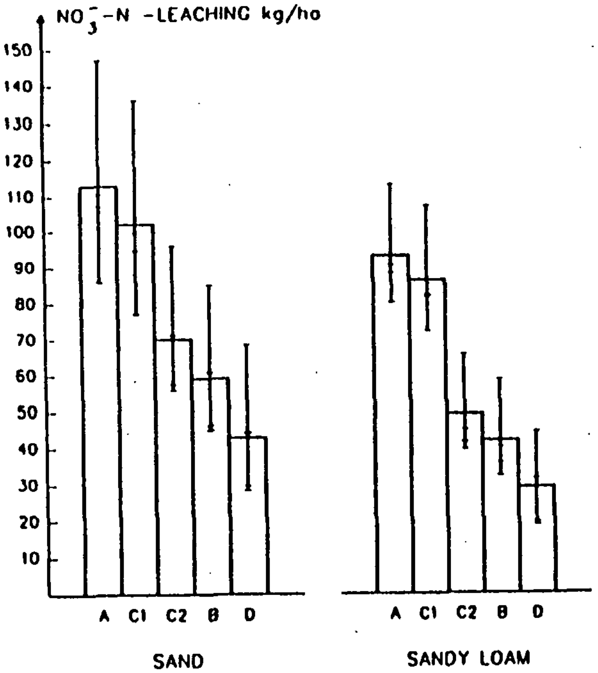

Figure 3 Leaching from a cropping sequence on sand and loamy sand, as a function of application time and fertilizer type, calculated as an average of 20 years. A, application of manure during autumn; B, application of manure during spring; C1 and C1, alternate application of manure during autumn and spring, with three and two autumn applications, respectively; D, application of mineral fertilizer only in the beginning of or in the growing period. (*Source:* Ref. 26.)

Not surprisingly, application of manure during autumn leads to serious leaching, while manure application during spring, close to the growing season of the plant, results in much less nitrate loss. Use of mineral fertilizer reduces leaching further. The bars in Figure 3 show the greatest and the smallest loss of the four repetitions of the 5-year sequence, while the columns show the average. The bars thus illustrates that there may be considerable difference in leaching from fields receiving identical treatments but subject to different weather conditions. Autumn or spring periods with high rainfall lead to increased leaching. However, very dry summers may also increase leaching because the plant uptake of nitrate is limited by water stress and thus more nitrate is subject to leaching after harvest. In this example, the sandy loam looses a little less (15–20 kg ha^{-1}) than the sand. This may be due to the fact that eastern Denmark receives approximately 100–150 mm less rainfall than western Denmark per year, and because the lower hydraulic conductivity of the sandy loam delays the percolation.

Another series of simulations illustrates that calculation (or measurement) of leaching from a given crop may be quite complicated. Three 5-year cropping sequences were constructed, consisting of a mix of winter wheat, barley, and sugar beet, the third crop in the sequence being the one to be investigated. Again each sequence was repeated four times, using 20 years of weather data, and the same soil types as in the above example. Each sequence was simulated with normal or slightly low fertilizer rates to crops no. 1, 2, 4, and 5 in the rotation, but with four different rates of fertilizer for crop no. 3. The difference in leaching among the four treatments were then calculated for the 20-year period, and divided by 4, to attribute an average leaching to each of the four repeated sequences. As might be expected, the effect of increased fertilizer rate for one crop in the sequence may be rather limited on the following crop on sand, but it may be considerable on sandy loam, due to less leaching of nitrate. After application of organic fertilizers on sandy loam, there may be an effect in both the second and third years after application. A single simulation, using barley undersown with grass instead of just barley, showed that the increased organic pool in the soil due to incorporated grass leads to increased leaching later in the sequence. Winter wheat, with a long growing season, decreases leaching compared to spring barley. For the beets, the effect of a relatively long growing period is overridden by the application of manure. Due to the fact that the effects later in the sequence are included in the calculation, the curves (Figure 4) representing the leaching appear straighter than is generally the case in leaching studies from the literature.

B. Example Two, Catchment-Scale Modeling

This example illustrates the combined use of the DAISY model and the MIKE SHE modeling system to simulate nitrate transport and transformation from application at the fields to the outflow in the river (or, more precisely, to the wetland areas). The model application was part of a large Danish research project [19,21,22] in which a number of specific studies related to nitrate pollution were carried out. One of the objectives of the modeling exercises was to assess nitrate transport and transformation on a regional scale using existing data. As a basis for the regional modeling, a number of process studies were carried out.

The area under study (Figure 5) was the 425-km^2 large Karup River catchment, located in the western part of Denmark. The aquifer consists of sand deposits with a thickness ranging from 20 to 100 m deposited by braided rivers. The aquifer is unconfined, with a water table below the surface ranging from [illegible] near the river system to 25 m near the eastern gr[illegible]dwater divide.

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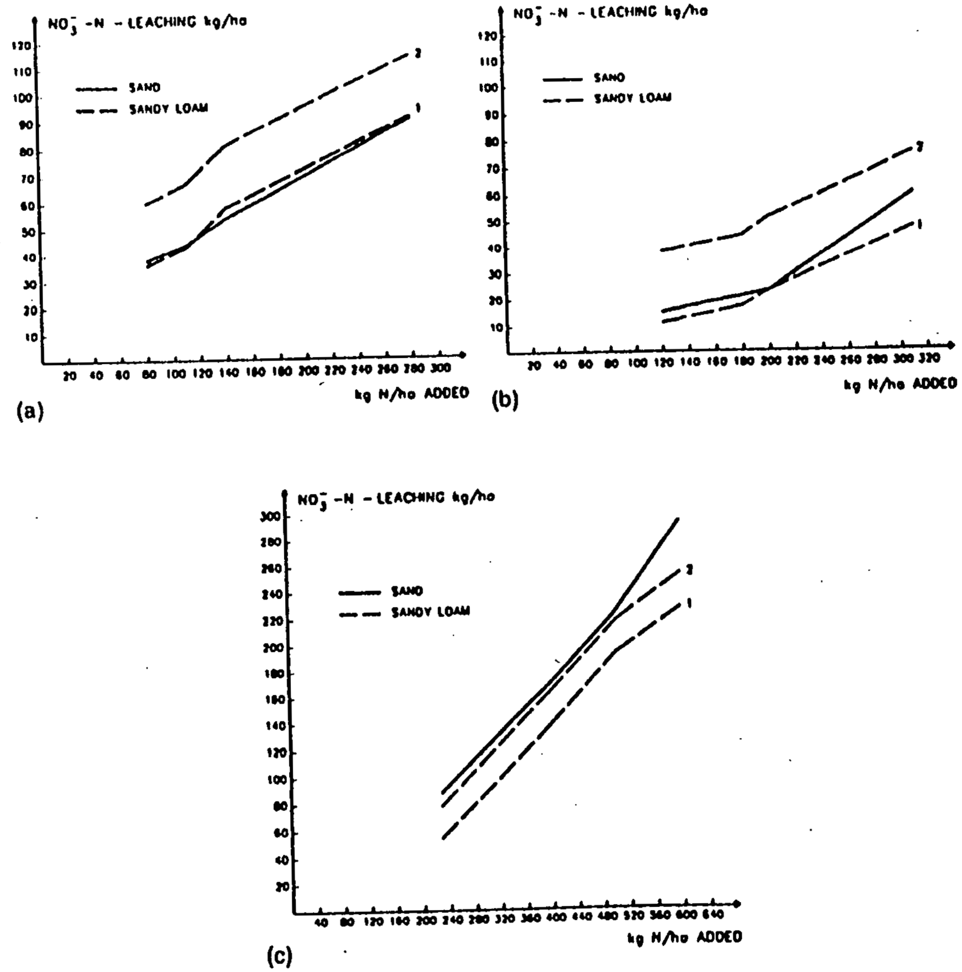

Figure 4 Leaching below (a) spring barley, (b) winter wheat, and (c) beets as function of fertilizer amount. The leaching is calculated for the calendar year (1) in which the crop is grown for sand and sandy loam. For sandy loam, leaching is also calculated for the period 1/5–30/4 (2), because percolation is slower on this soil type. For example, leaching of nitrate due to mineralization of winter wheat residues during autumn may not show up at 1.2 m (leaching depth in this study) before the following spring. (*Source:* Ref. 26.)

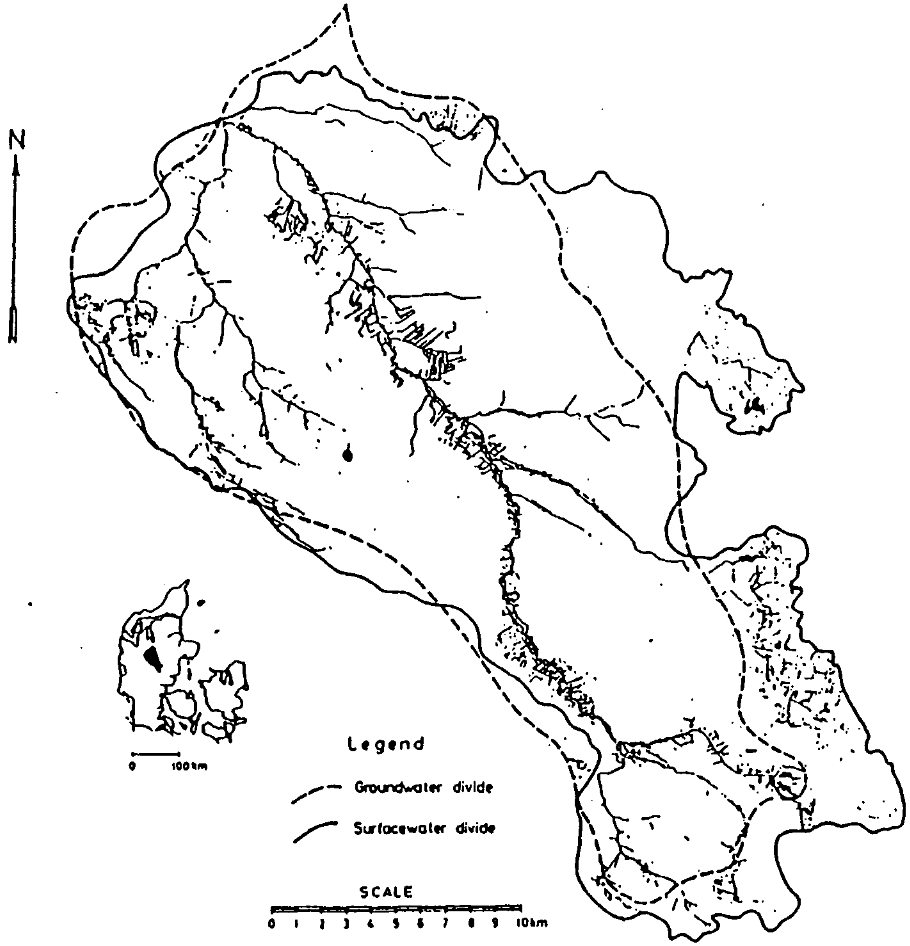

Figure 5 The Karup River basin and drainage pattern. Both groundwater and surface water divides are shown. (*Source:* Ref. 30.)

The area is partly arable land and partly covered by forest and heather. A major part of the riverine zone along the main river has been drained and is today converted into cultivated land.

Detailed monitoring of the nitrate concentration in the groundwater aquifer was carried out in a 10-km² subcatchment of the Karup catchment. This study provided the justification for assuming the simple reactive nitrate reduction model [27].

Figure 6 illustrates schematically the coupling between the DAISY

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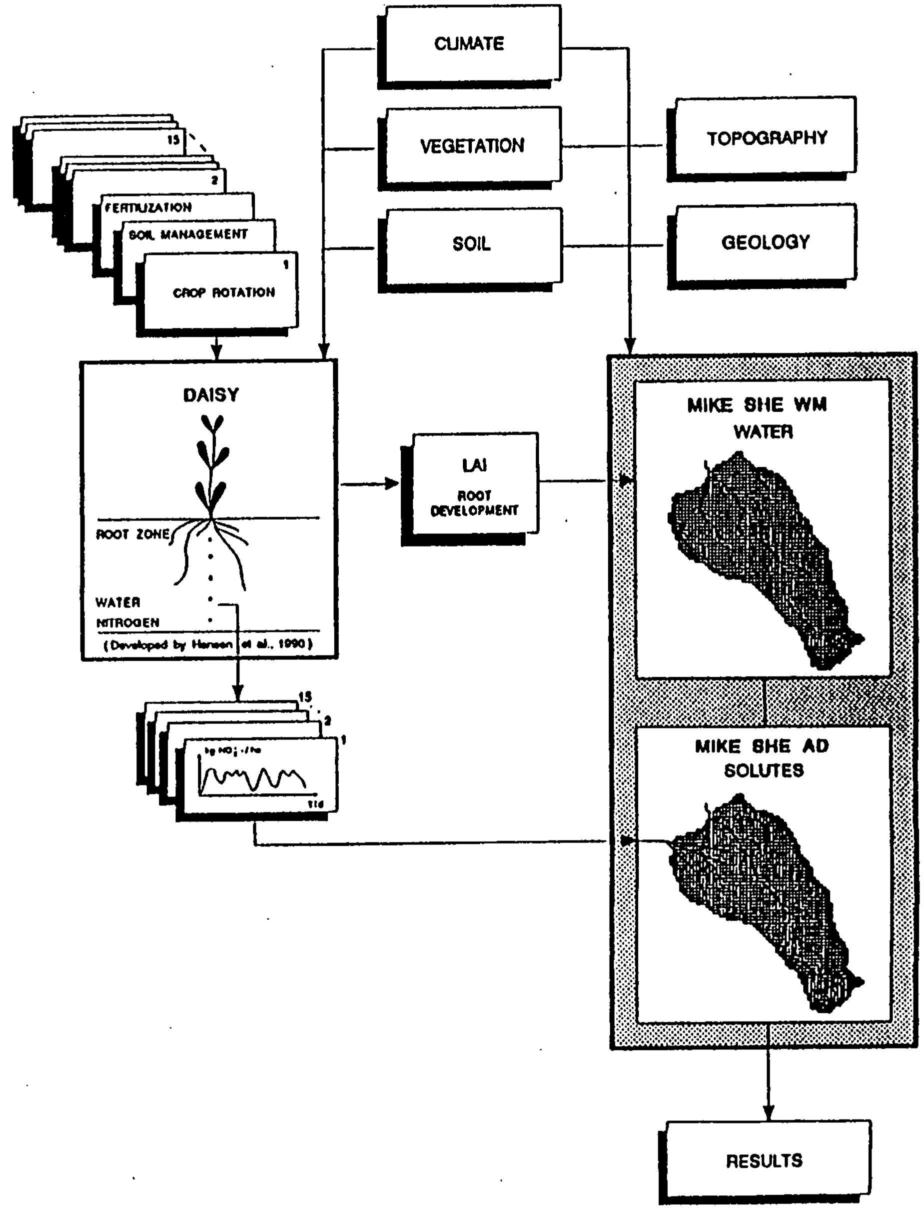
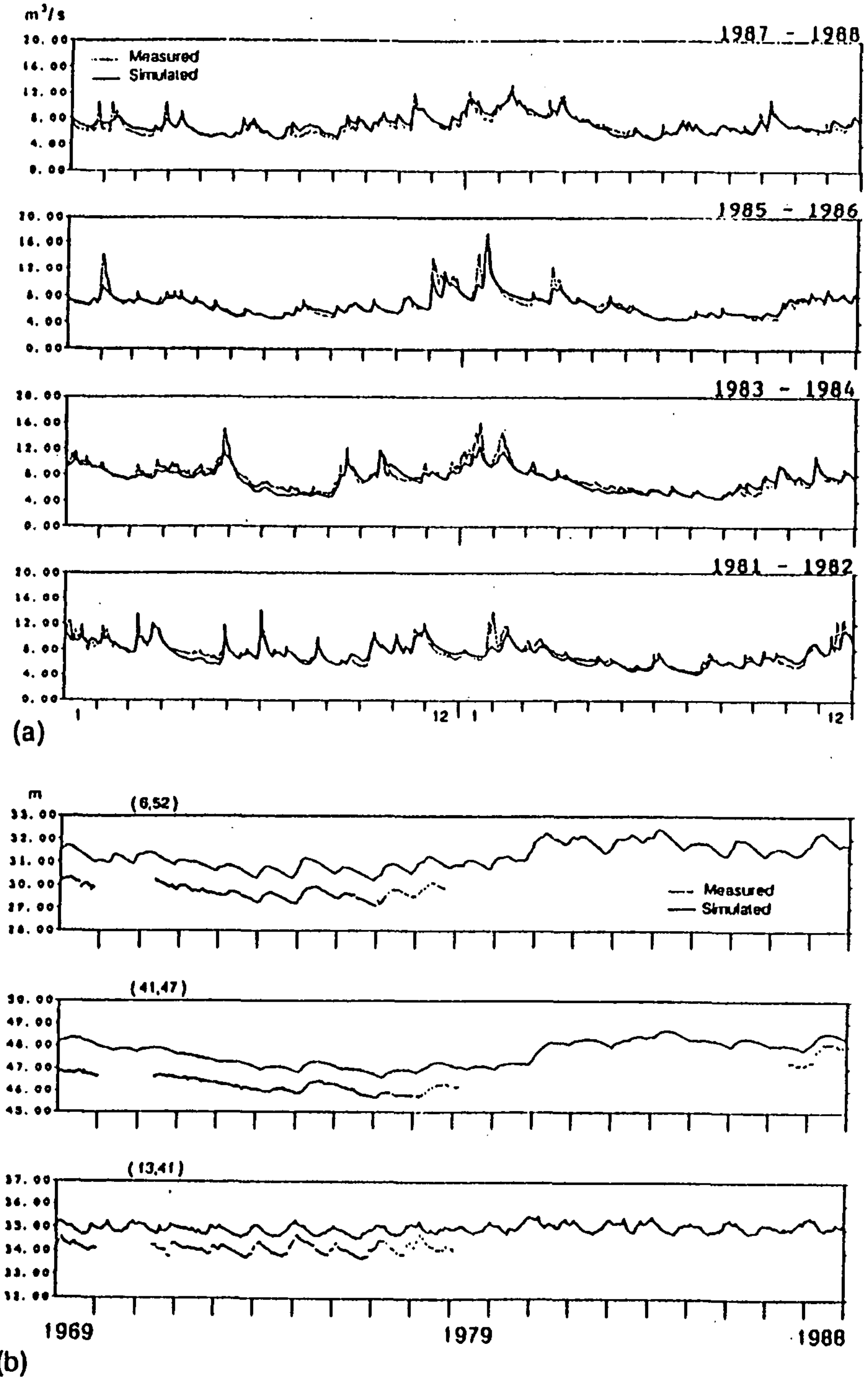


Figure 6 Schematic representation of the data requirements for the two models and the linkage between them. (*Source:* Ref. 21.)



model and the hydrological distributed modeling system MIKE SHE. The following modeling procedure was adopted:

DAISY was used to simulate time series of nitrate leaching from the root zone for 15 defined cropping sequences covering 20 years. The cropping sequences were attached to a number of elements in the horizontal grid network; in total, they represented the observed spatial distribution of the crops at any time. The distribution of the individual crop sequences is done partly at random. However, areas with permanent grass are located mainly in the areas along the river system, and as the southern part of the catchment is known to include most of the pig farms, these were located mainly in this area.

MIKE SHE was calibrated against observed groundwater heads and river flows to simulate the water flow in the catchment as correctly as possible. For the calculation of the evapotranspiration rates, the crop development parameters simulated with DAISY were used.

The computed leaching rates were used as input to the solute transport module of MIKE SHE, which simulated the advection and dispersion in the unsaturated zone and the groundwater zone. The nitrate was simulated as a conservative solute in the unsaturated zone below the root zone. In the saturated zone all nitrate carried across the redoxcline was reduced.

The collection and processing of the input data for the DAISY and MIKE SHE are described in detail in Ref. 22. Assessment of the fertilizer applications and agricultural practices was carried out for the period 1968–1988 from statistical data and information from agricultural advisers. For the same period, data was available for calibrating the water flow component of MIKE SHE (Figure 7).

Figure 8 shows the simulated nitrate concentrations in the upper part of the groundwater in six grid squares inside the model area. The figure shows at the same time the pronounced the effects of both farming practice as well as the effect of the unsaturated zone on the simulated nitrate concentrations in the upper part of the groundwater zone. Simulated variations in nitrate concentration are larger and reflect the application rates more closely in areas with a shallow water table compared to zones far away from the river system with a larger distance to the groundwater. This is important to realise, because management options may have different effects depending on the location.

Figure 7 Comparison of simulated (a) discharge and (b) groundwater heads with corresponding measured values. The figures in brackets are grid references. (*Source:* Ref. 22.)

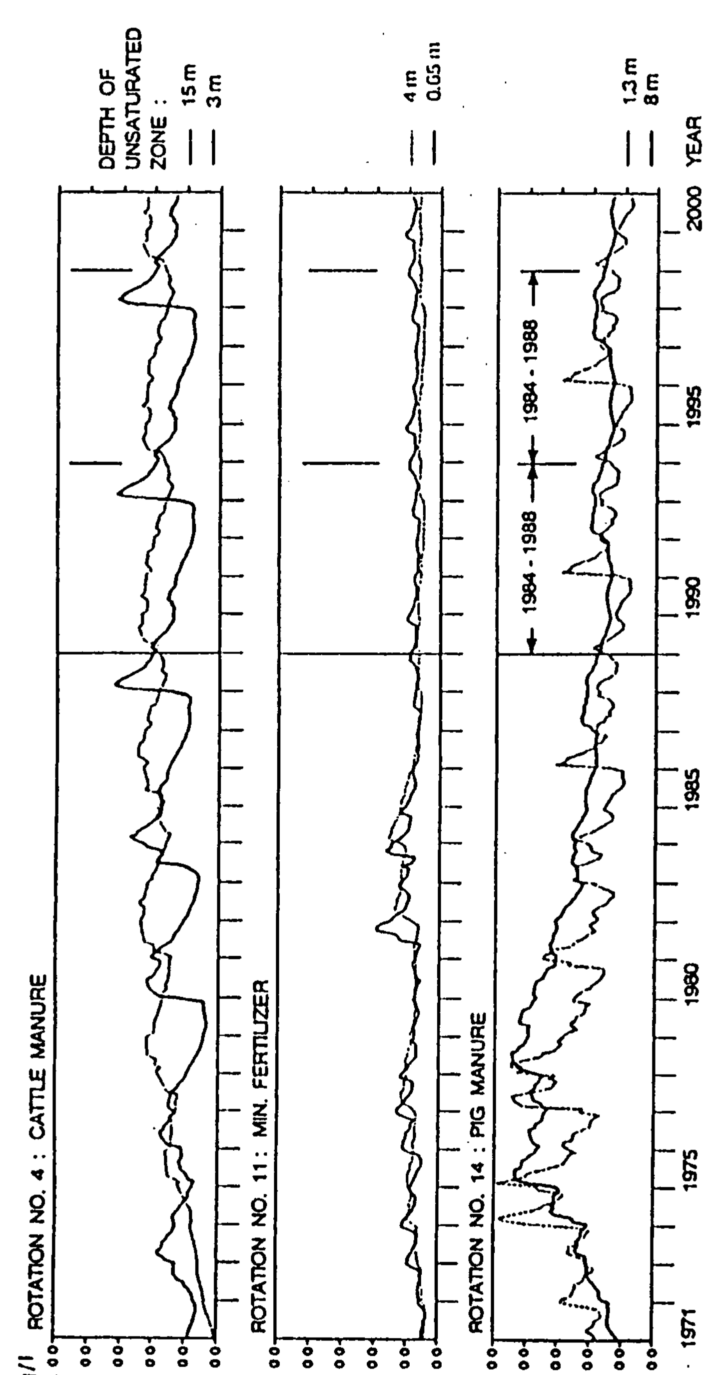


Figure 8 Temporal variation in NO_3^- concentrations in the upper groundwater layer beneath three selected cropping sequences, with two different distances to the groundwater table. The data are extracted from selected grids. (*Source:* Ref. 22.)

It should be noted that the simulated values cannot be validated, but they compare well with the general level and amplitudes found in the area [19,22].

Another important analysis in connection with management decisions for surface waters, rivers and lakes, is to calculate the time lag between addition of nitrate to the arable part of the catchment and its appearance in the river system (wetland areas). Figure 9 illustrates the predicted effect of a pulse leaching from the root zone uniformly distributed over the agricultural area. It is seen that after 3 years, 30% of the nitrate has seeped into the wetland areas, whereas 45% has been reduced in the aquifer. After 15 years, half of the leached nitrate has entered the wetland areas.

Thus, Figure 9 provides information about how quickly measurable changes can be expected in surface and groundwater after a change in management of the agricultural areas, and it translates the quantitative change in input to a quantitative change in output.

Comparisons with observed nitrate concentration in the Karup River showed that approximately 50% of the nitrate entering the wetland areas is reduced. This figure compares well with an independent field study undertaken in the area [1].

Several future scenarios were tested for the Karup River catchment, including changes in timing of manuring, changed distribution of the manure between fields, and maximization of cropping sequences with winter crops to ensure that the ground is covered for as large a part of the year as possible.

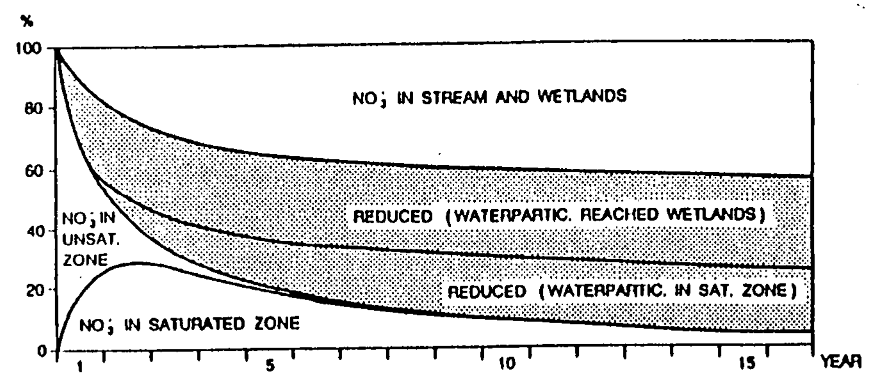


Figure 9 Relative distribution of nitrate after a single addition, equally distributed over the agricultural areas in the Karup stream catchment. The reduced nitrate is divided into two parts, one which would have reached surface waters and one which would still have been in the groundwater, had it not been reduced. (*Source:* Ref. 22.)

It was concluded that such changes would reduce the nitrate losses considerably, but not quite enough to reach the level prescribed in the Danish Environmental Plan (50% of 1980–1984 level).

An important limitation in the result is that the DAISY/MIKE SHE model approach did not include denitrification processes in the wetland areas. There are two reasons for this. First, although the horizontal discretization (division into subareas) may seem detailed, a proper description of the conditions in the wetland areas cannot be provided by a regional model. It requires a much finer discretization to account for the local variations in drainage pattern and extend of wetland zone. Second, at the time of the study, a wetland model such as the one developed by Dørge [25] was not available.

With the present stage of model developments (1993), a range of alternative scenarios can be analyzed. These include a range of changes in management on the fields, but also effects of introducing buffer zones with fallow and/or reestablishment of the natural wetland conditions in previously drained areas along the river system. It would also be possible to determine in which areas along the river system such changes would have the greatest effects.

VII. CONCLUSION

Case studies have shown that it is possible to put together models to be used for analyses of management options for point, field, and catchment scales. Models are now available for almost all phases of the system. Correctly selected and used, such models must be regarded as the strongest tool available for analyses of management options.

To carry out advanced and integrated analyses requires expertise in several fields. The parameter estimation, the calibration phase, and the interpretation of the results are best done by a team of experts/scientists with complementary backgrounds.

Data availability may be a limitation on particularly regional analyses, It may, however be less serious than expected due to possibilities of using statistical information, etc. Also, a team of experts is likely to be able to collect and interpret a much larger selection of databases than experts from one field only.

It should be realized that, particularly for the regional modeling exercises, the results are not independent of the modelers, as they act as screeners and interpreters of data.

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Nitrate and Mn-chemistry in the alluvial Danubian Lowland aquifer, Slovakia

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Abstract The biogeochemical processes associated with infiltration of the Danube River water in a coarse, slightly reactive aquifer are studied. Groundwater quality data show slow denitrification and reductive dissolution of Mn-oxides, possibly manganite. Whether or not these processes interact is not known. A simple kinetic denitrification module is coupled to an equilibrium-controlled geochemical multi-component transport model. The preliminary modelling results show that the general patterns can be reproduced, however, no detailed modelling is yet possible. This requires better knowledge on the kinetic parameters, among which the amount of metabolizable organic matter.

INTRODUCTION

Alluvial basins amidst mountain ranges are of strategic importance as a drinking water resource. The alluvial aquifer may be recharged by the river as is the case for the Danubian Lowland basin, Slovakia. The geochemical processes associated with river water infiltration can be relevant for present and future groundwater quality. The purposes of this study are: (a) to investigate the ongoing (bio)geochemical processes during infiltration of Danube river water into the aquifer, (b) to develop a numerical (bio)geochemical transport model that describes the following phenomena: advection/dispersion, microbial degradation and inorganic equilibrium geochemistry, and (c) to model the groundwater quality observed in the field.

HYDROGEOLOGICAL SETTING AND FIELD INVESTIGATIONS

The study area lies in the Danubian Lowland. This is an inland delta formed in the past by river sediments from the Danube and is situated between Bratislava and Komárno, southwestern Slovakia. The entire area forms an alluvial aquifer, which throughout the year receives approximately 25 $m^3 s^{-1}$ infiltration from the Danube River in the upper

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parts of the area and returns it into the Danube River and drainage channels in the downstream part. The groundwater flow rate in the immediate vicinity of the Danube River is high, a few metres per day. In addition to recharge from the Danube River, a small amount of recharge originates from land surface. The alluvial aquifer primarily consists of gravels and coarse sands with some intercalated fine meander deposits and sands. The aquifer is overlain by a loamy loess deposit of 1 to 3 m thick. The aquifer is an important water resource for municipal and agricultural water supply.

Multi-screen wells were installed in the upstream part of the Danubian Lowland just south of Bratislava, in a cross section that is 7.5 km long parallel to the general groundwater flow direction. The distance between wells increases with increasing distance from the Danube River. It must be realised that the Danube River cannot be considered as a line source for infiltrated groundwater due to the large width of the Danube River: the travel times to the different screens of the first well may vary significantly.

RESULTS AND DISCUSSION OF FIELD INVESTIGATION

The groundwater quality data show that different types of groundwater can be recognized with respect to origin and redox-status (Fig. 1(a)). Surface-recharge groundwater is found in shallow inland screens, in addition to river-recharge groundwater. The distinction can best be made by differences in SO_4, alkalinity and sometimes also NO_3. Concentrations larger than the average Danube River water concentration plus one standard deviation are attributed to surface-recharge water. The transition between the two types lies somewhere between well 7 and well 9.

Oxic groundwater (O_2 > 3 μmol l^{-1}) is found in shallow screens of wells 3 through 11. Nitrite is also frequently found above detection limit (0.4 μmol l^{-1}) in non-oxic groundwater, which indicates the occurrence of denitrification in a porous medium that has a low reactivity for organic matter. Further, high Mn-concentrations (approximately 20 μmol l^{-1}) are found in a zone ranging from wells 1 through 7 in intermittent screens. Intermediate Mn-concentrations (1-10 μmol l^{-1}) are found in association with this high Mn-zone and in several screens of wells 10 and 11. River-recharged groundwater having high or intermediate Mn-concentrations was always O_2-depleted (O_2 < 3 μmol l^{-1}), whereas surface-recharge groundwater having intermediate Mn-concentrations was also O_2-bearing. The Mn-concentrations are rather constant in the observation period from April 1993 to July 1994 and lie above the Danube River concentration for several screens, which indicates that Mn is mobilized after infiltration.

The pattern for NO_3 is more complicated. The Danube River has a seasonal fluctuation with usually a minimum in July-September (0.08-0.13 mmol l^{-1}) and a maximum in March (0.25-0.40 mmol l^{-1}). Groundwater in the vicinity of the Danube River also shows seasonal fluctuation with a delay compared to the Danube River. Some screens show concentrations that suggest denitrification (e.g. 1.2, 1.3, 2.5; see Fig. 1 for notation of screen numbers), whereas for other wells no indications for denitrification are found (e.g. 1.4, 1.5, 2.2 to 2.4). A spatial pattern cannot be recognized. Neither is a pattern found that coincides with the occurrence of either Mn or O_2. The concentrations for the inland screens that reflect river-recharged groundwater are somewhat lower than the range for the Danube River water. The decrease in NO_3 concentration due to denitrification is estimated to be 0.03 to 0.15 mmol NO_3 l^{-1}.

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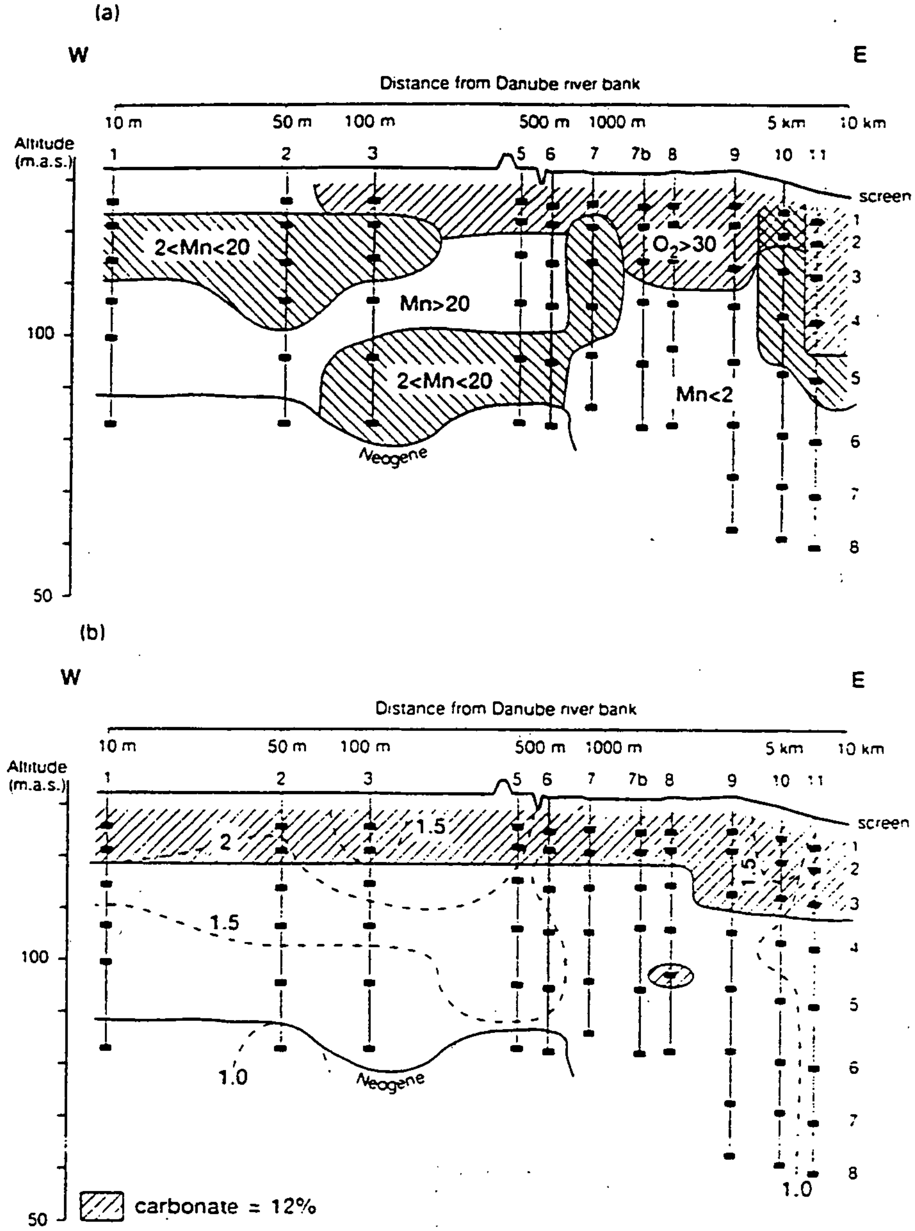

Fig. 1 (a) occurrence of O_2 and Mn (μmol l^{-1}), (b) occurrence of carbonate and DOC (mg C l^{-1}).

The concentrations of DOC, COD_{Mn} and COD_{Cr} decrease from the most shallow zone close to the Danube River in deeper and more western direction (Fig. 1(b)). Remarkably, this trend does not coincide with the patterns found for O_2 and Mn, hence the distribution between oxic, suboxic/Mn-rich and suboxic/Mn-poor groundwater.

The act of denitrification together with the behaviour of Mn deserve attention. Thermodynamically, Mn and NO_3 are unstable in their mutual presence at pH is

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approximately 7.5 for the concentrations observed. This means that the changes in groundwater composition must be kinetically controlled. An examination was performed of the influence of the redox potential on the saturation indices for Mn-minerals using WATEQ4F (Ball & Nordstrom, 1991). The redox potential was calculated from the O_2 concentration for the oxic groundwater types. The groundwater was always supersaturated for Mn-oxyhydroxides and unsaturated for rhodochrosite ($MnCO_3$). A somewhat different picture is obtained for the non-oxic water types. The saturation index for rhodochrosite ranges between −1.0 for low Mn concentrations to 0.5 for high concentrations. The individual values calculated are largely directly proportional to the Mn concentration and the values are independent of the redox potential as long as the potential is outside the range of oxic water. Rhodochrosite can thus not be the unique source for Mn, since no supersaturation for rhodochrosite would then be found and, additionally, no major increases in the coion, HCO_3, or pH are observed which might result in supersaturation.

The values for the Mn-oxyhydroxides are of course very sensitive to the redox potential used. It is of utmost interest that when the redox potential is calculated from the NO_2/NO_3 couple, the saturation index for manganite, Mn(III)OOH, is mostly within one unit from saturation. The average was 0.04 and its standard deviation was 0.65 for all NO_2-bearing samples, being either Mn-rich or Mn-poor. Unsaturation was always calculated for other Mn-hydroxides. This suggests an active role for manganite, or more generally Mn-oxides, as source. Precipitation of rhodochrosite or cation-exchange with aqueous Mn can act as a potential sink following mobilization of Mn by dissolution. The origin of Mn from manganite dissolution implies that a reductor for Mn(III) needs to be found, since Mn occurs in its divalent state in groundwater. Reductive dissolution of Mn-oxides may happen in two ways: direct and indirect (Nealson *et al.*, 1989). Manganese itself is involved in the microbially mediated redox process in direct mechanisms, whereas in indirect pathways it is indirectly reduced by means of a product of the microbially mediated process. In case of indirect cycling, NO_3 reduction can be assumed to mediate Mn(III)-dissolution, where one of the products of denitrification

Table 1 Reaction budgets for different reactants in the geochemical cross section. Negative values refer to oxidants and positive ones to reductants. Distribution of carbonate is presented in Fig. 1(b).

	River-recharged groundwater:		Aquifer sediment:	
	(mmol l^{-1})	(melec l^{-1}_{gw})	(melec l^{-1}_{gw})	(ppm $_{<2\ mm}$)
O_2	0.31	−1.25		
NO_3	0.08	−0.40		
Mn(II)	0.018	0.018		
DOC	0.15	0.60		
$TOC_{carb\text{-}rich}$			1780 ± 2070	4400 ± 3500
$Mn(III)_{carb\text{-}rich}$			−0.9 ± 0.5	38 ± 9
$TOC_{carb\text{-}poor}$			256 ± 193	800 ± 600
$Mn(III)_{carb\text{-}poor}$			−1.96 ± 2.03	143 ± 256

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(NO_2^-, NO, N_2O or N_2) is considered to cause reduction of Mn(III). Similar processes are known for coupling of denitrification with Fe-redox cycling (Brons, 1992).

A comparison of reaction budgets is made to illustrate the relevance of the different reactions. Table 1 presents the changes in concentration and contents analysed for the relevant species. The sediment analyses are converted into electron budgets per volume groundwater for redox processes assuming a porosity of 33% and a bulk density of 2.65 g cm^{-3} and taking into account the gravel fraction. The calculations suggest that DOC supplied from the Danube River, may act as reductant in the groundwater aquifer. However, it cannot be the only reductant, since the amount is much smaller than the amount of reduced O_2 and NO_3. Solid organic matter must also be a reductant. The electron budgets calculated are 1.96 ($\pm$2.03) and 256 ($\pm$193) melec $l^{-1}{}_{gw}$ (groundwater) for Mn-oxides and TOC, respectively. Here, the Mn-content associated with oxides was determined by selective extraction of oxides using ascorbic acid/oxalic acid after removal of carbonates (Griffioen & Broers, 1993) and it is assumed that Mn is in its trivalent state. The budget for Mn-reduction is clearly much smaller than for organic matter oxidation. The budget for solid Mn(III) exceeds the aqueous Mn-concentration approximately a factor of 100. The budget for solid organic matter is much larger than for aqueous organic matter, but the reactivity may differ considerably.

It is not clear whether denitrification and reductive dissolution of Mn-oxides interact or not. It can be expected from thermodynamics that the presence of approximately 0.15 mmol NO_3 l^{-1} competes out microbial reductive dissolution of Mn-oxides. It can be argued from microbiological and kinetic points of view that reductive dissolution of manganite can only be expected for dissolved organic carbon as reductor, since a reaction between a solid reductor (organic C) and a solid oxidant (Mn-oxides) cannot be catalysed by bacteria. Such limitation does not happen for denitrification. The present data for screen 1.5 suggest, for example, that reductive dissolution of Mn may happen before any denitrification has started, whereas for screen 1.3 little reductive Mn-dissolution has happened while denitrification is an ongoing process. The mobilization of Mn seems limited to a restricted zone. However, no such zone is indicated for denitrification, whereas for O_2-depletion a spatial distribution is also found.

More generally, the occurrence of the redox processes may depend on the geohydrological and geochemical heterogeneity of the system. The next two sections present the modelling approach and results for coupling transport, microbial degradation and inorganic equilibrium thermodynamics to study in a general way the (bio)geochemical processes. The particular question is whether the manganese/nitrogen redox cycle can be considered as abiotically, thermodynamically controlled and the nitrogen/carbon redox cycle as biotically, kinetically controlled.

NUMERICAL MODELLING APPROACH

The numerical model consists of three components: (a) advective-dispersive transport of all dissolved components, (b) kinetically-controlled denitrification, and (c) (pseudo) equilibrium-controlled speciation and equilibrium reductive dissolution of manganite. Although both organic matter in a dissolved and a solid state are believed to contribute to the reduction capacity, only that in the solid phase has been included at present. The calculations within one time step can be divided into four steps. The MIKE SHE family

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of models, the MIKE SHE AD model (DHI, 1993) and the GEOCHEMISTRY and BIODEGRADATION models (VKI, 1994), have been used to develop the model.

First step is transport of the total aqueous component concentrations (u_j) and the total redox state (R_s). Both represent a mass balance in the dissolved phase for the components and the "available" changes in oxidation states. Engesgaard & Kipp (1992) have presented the mathematical derivation for the transport equations for both u_j and R_s. Second step consists of an equilibrium speciation at all grid points. The speciation is based on u_j, R_s and using a specified thermodynamic data base. Here, it is of particular interest to know the individual concentrations of NO_3 and NO_2.

In step three, denitrification is simulated at all grid points. The field and laboratory investigations have not directly identified and quantified the reaction pathways in the denitrification process. Consequently, it has been decided to simulate the oxidation of organic matter in a rather simplistic manner. It is assumed that a single reaction occurs on an electron basis, between organic matter and "nitrogen oxidation capacity" called $R_{s,N}$. The capacity is defined as

$$R_{s,N} = 5[NO_3^-] + 3[NO_2^-] - 0[N_2] \tag{1}$$

where $[i]$ refers to concentration of solute i and the coefficients indicate the oxidation state for the species of interest. The $R_{s,N}$ is therefore just a fraction of the total redox state. It is moreover assumed that the reaction follows a first order rate law with respect to nitrogen:

$$\frac{dR_{s,N}}{dt} = -\mu_N R_{s,N} \tag{2}$$

where μ_N is a first-order utilization rate [T^{-1}]. Equation (2) therefore accounts for the decrease in "available" changes in oxidation states for the nitrogen system. The calculated decrease in $R_{s,N}$ can simply be related to a decrease in CH_2O if the content of soil organic matter is based on units of electrons per aqueous volume:

$$\frac{d[CH_2O]}{dt} = \frac{dR_{s,N}}{dt} \tag{3}$$

The model for microbial denitrification does not distinguish between the individual nitrogen species, rather, they are all treated together through the lump parameter $R_{s,N}$. The total nitrogen concentration remains constant during denitrification.

In the fourth step, the change in the redox state of the nitrogen system is allowed to affect aqueous speciation and mineral equilibrium, through the definition of a new total redox state, i.e.

$$R_s^F = R_s^I - \Delta R_{s,N} \tag{4}$$

where R_s^I and R_s^F are the total redox states before and after denitrification, respectively, and $\Delta R_{s,N}$ is the calculated decrease in nitrogen oxidation capacity. The change in the total redox state will cause a state of non-equilibrium with respect to manganite thus triggering reductive dissolution of manganite. It is here assumed that reductive dissolution of manganite can be described as an equilibrium process. Reduction of Mn(III) in manganite to aqueous Mn(II) must be associated with the oxidation of a reduced nitrogen species. If thermodynamic equilibrium constants are used for the

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nitrogen system. NO_2 will never show up in groundwater (Stumm & Morgan, 1981). To reflect the kinetics of the denitrification process, the equilibrium constant for the N_2/NO_3 half-cell reaction

$$2NO_3^- + 10e^- + 12H^+ \rightleftharpoons N_2 + 6H_2O \qquad (5)$$

has been empirically adjusted to obtain a situation, where NO_2 is calculated as an intermediate product. The complete model thus possesses two semi-empirical parameters: the rate parameter μ_N determines the rate of denitrification and the apparent constant $\log K_{N_2/NO_3}$ determines the interaction between denitrification and reductive dissolution of Mn-oxides.

MODELLING RESULTS

A sequence of modelling scenarios was undertaken in a simple generic, uniform, steady-state flow system in order to demonstrate the simulation capabilities of the model. The flow system represents a cross section of 80 m long and 11 m deep. A "Danube River" source was introduced in the middle of the cross section at the upstream end. The source and initial compositions were constructed from the analysis of an actual river water sample in the following way. The source is thought to represent the conditions below the river bed, where all O_2 is quickly consumed by (dissolved) organic carbon. The redox potential is adjusted until the concentration of O_2 was practically zero while making sure that all nitrogen is present as NO_3. The source R_s is calculated to be 15.54 meq l^{-1}. The exact same conditions were used for the initial aquifer conditions, except that the total nitrogen concentration is set to zero. This gives an initial R_s of 14.78 meq l^{-1}. The total Mn concentration is very low initially in the aquifer. The two solutions only differ in their total nitrogen concentration and, concomitantly, the total redox state. Setting up the geochemical system in this fashion makes it more easy to reflect the ongoing geochemical processes.

The influence of μ_N on the biogeochemistry was investigated through a series of simulations that reflect no, low, medium, high, and very high rate of denitrification corresponding to rate values of 0.0, 0.0007, 0.007, 0.02, and 0.07 day^{-1}. Through experimenting with the $\log K_{N_2/NO_3}$ of reaction (5), it was found that a value of approximately 160 instead of 207 can give the correct concentration of NO_2 in the system with a low to high denitrification rate. The simulations are summarized in Fig. 2, which shows breakthrough curves approximately in the middle of the cross section. With no denitrification, nitrate is transported conservatively. Including denitrification with a low rate, results in the consumption of small amounts of NO_3 and the appearance of NO_2, but no N_2. Nitrite breaks through after NO_3. Notice that the NO_2 concentrations are much smaller than the NO_3 concentrations. With an increase in the rate of denitrification to medium and high, no NO_3 appears, because it has already been degraded. Instead NO_2 breaks through earlier displaying a peak, where the peak value depends on the rate. With a very high rate of denitrification, no NO_2 will appear, and all NO_3 is directly converted to N_2.

The results are consistent with the interpretation of the microbial denitrification process. All these results are a function of the chosen $\log K_{N_2/NO_3}$ for reaction (5). If

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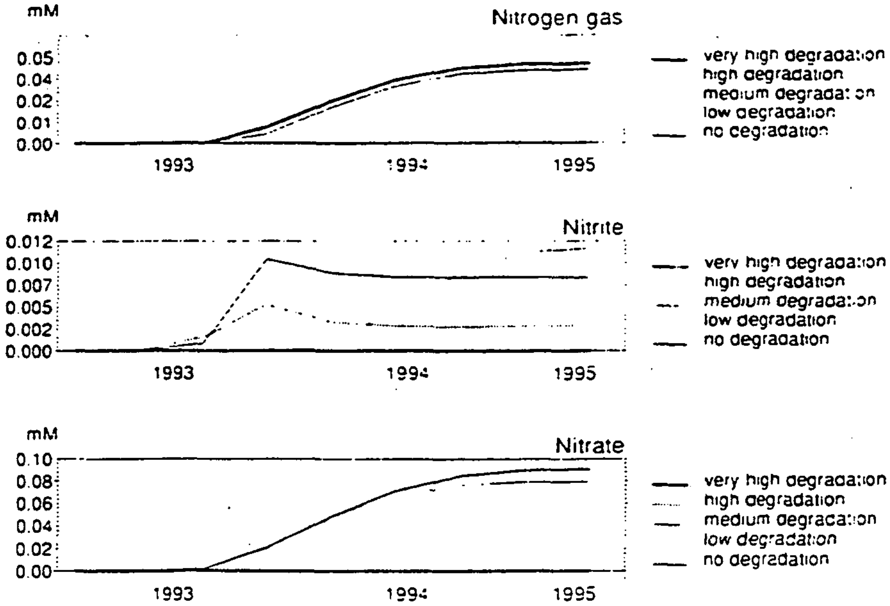

Fig. 2 Breakthrough curves of N-species for different degradation constants, using an adjusted value for $\log K_{N_2/NO_3}$.

another value will be used then the rates will be different, but the relative changes in rates will still affect the results in much the same manner as described above. The proposed model can thus be considered as a simple and useful denitrification model. The model is used to simulate transport and denitrification at the field site. The modelling is still ongoing and the results presented here are preliminary. The objective has been to demonstrate that with a choice of μ_N and $\log K_{N_2/NO_3}$ it is possible to qualitatively describe the observed distributions for the nitrogen and manganese species.

The model was set up for geohydrological boundary conditions that are valid for the Danubian Lowland aquifer. Denitrification in the actual Danube River cross section was modelled using the source and initial concentrations as described above. Figure 3 shows that the NO_2 plume happens within the zone of actual reduction of NO_3. Manganese is solubilized in the same zone as NO_2 but is transported more downstream since it remains in solution whereas NO_2 is degraded. These results briefly show the biogeochemical phenomena of the model developed. More detailed modelling is necessary. Here, the following factors need to be considered for the act of the two reactions: (a) availability of metabolizable organic carbon, either in a solid or a dissolved state, (b) availability of Mn-oxides, (c) reaction rates, (d) groundwater flow rate, (e) type of river water bottom and, possibly, (f) temperature.

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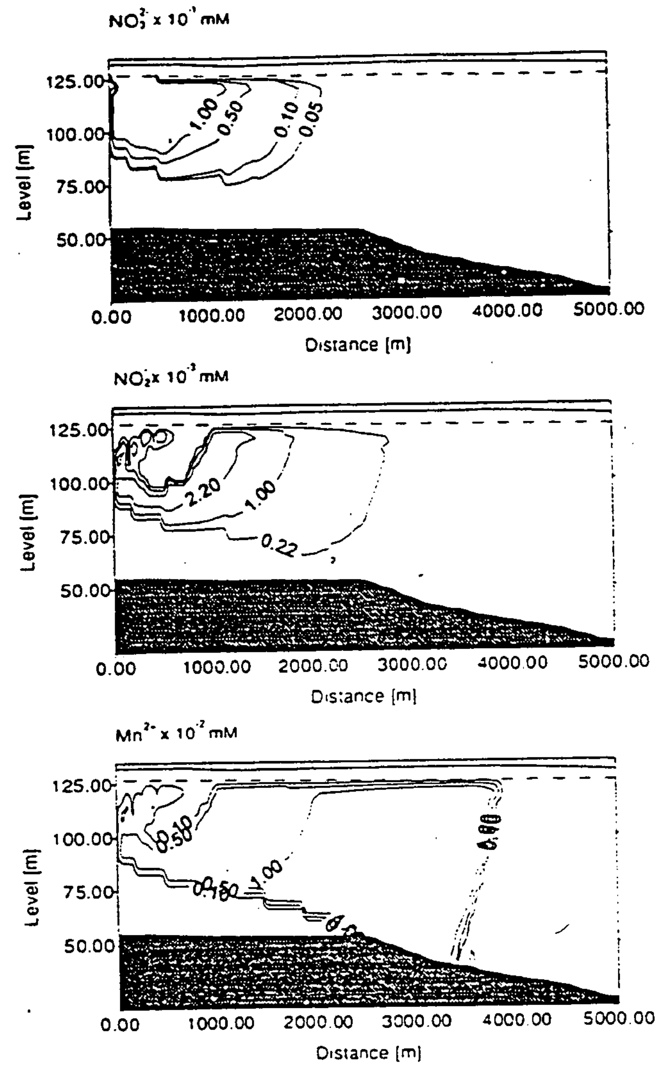

Fig. 3 Modelled distribution of NO_3, NO_2 and Mn in the Danubian Lowland aquifer for a coupled N/Mn redox system after 4 years of injection of NO_3-bearing the Danube River water.

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SKRYDSTRUP WASTE DISPOSAL SITE - A CASE STUDY

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ABSTRACT

The Skrydstrup waste disposal site and the unconfined aquifer downstream placed in the Southern Part of the peninsula Jutland, Denmark, is one of the most intensively investigated locations in Denmark. Deposits consist of a mixture of municipal waste, construction materials and chemical waste from a nearby refrigerator factory i.e. chlorinated hydrocarbon solvents and organic phosphorus components mostly stored in barrels. In 1986 it was discovered that a large part of the aquifer was polluted with both 1,1,1-trichloroethane, tri-chloroethylene and other chlorinated aliphatics and a pump-and-treat system has been in operation since 1988 in order to remediate the pollution and prevent further spreading.

However, in 1991 detailed studies of the geological and hydrogeological conditions as well as the contamination showed that the pollution had spread as far as two kilometres downstream the waste dump and approached the final recipient - a small stream. A three-dimensional groundwater and solute transport model was constructed on the basis of the new investigations and the conclusions based on the model simulations were that 1) the remediation will have no significant effect on the future spreading of the pollution plume and will certainly not prevent the plume from leaching into the recipient located about two kilometres downstream the waste disposal site; 2) even with 3.5 times larger capacity of the water treatment plant i.e. a total abstraction of about 110 m^3/hour and a more optimal location of the remediation wells it is not possible to recover more than 50% of the present contamination in the aquifer; 3) the amount of contamination leaching to the stream within the next 10 years is almost independent of the remediation scheme adopted and 4) more than 95% of the contaminated water will discharge to the stream within a distance of one and a half kilometre of the stream.

Combined with results from the investigations of the potential microbiological degradation of the contaminants in the area along the stream these findings lead to the conclusion that the pump-and-treat system was closed down and the effort in the coming years will be concentrated on control measures. Special focus will be given to the riparian areas along the recipient and the recipient it self.

Keywords: pump-and-treat, mathematical modelling, microbiological degradation potential, control measures.

INTRODUCTION

It is often proven that remediation of contaminated groundwater aquifers is a difficult task and optimization of pump-and-treat systems in order to obtain a certain groundwater quality with in a certain period often fails. This fact is to a large extent due to the unknown complexity of the geology and hydrogeology in the area under consideration as well as the chemical and physical properties of the contaminants (Mackay and Cherry, 1989, and Haley et al., 1991).

Geological and hydrogeological information is primarily obtained from measurements in pumping and observation wells. Since the number of observation points is limited, the available fragmental information has to be interpolated in time and space. This introduction of uncertainty in the description of the hydrogeological conditions of the aquifers is one of the largest limitation for carrying out reliable computer modelling of the solute transport processes and the remediation measures. On the other hand mathematical models are useful tools for compiling all available information in a consistent manner since the application and calibration of a model provides a method for successive confrontation of hypotheses and interpretations of the system and available measurements. This procedure will ultimately result in a tool which integrates all available information in an optimal way and mathematical modelling is the only way to effectively investigate and optimize location and pumping rates for remediation wells in a pump-and-treat system.

To be presented at the WEFTEC'95 Conference, October 21-25, 1995 at Miami Beach

Limited success with pump-and-treat systems which aim to extract the contamination from aquifers has lead to the conclusion that such remediation systems are more applicable for **controlling** the future spreading (Bredehoeft, 1992 and Nyer, 1993). However, the conclusion that an established pump-and-treat system which has been running for five years is so inefficient that it should be closed down has to the authors knowledge still only been seen at a limited (though incerasing) number of sites.

Project Background

Deposition of a mixture of municipal waste, construction materials and chemical waste at Skrydstrup waste disposal site took place in the period 1964 to 1974. The chemical waste originated from a nearby refrigerator factory and consisted mostly of barrels with paint and solvents for cleaning metal surfaces before lacquering. The waste disposal is located in an old gravel pit just above the water table of an unconfined sand aquifer.

Investigations in the area began back in 1984 and a severe contamination mainly with chlorinated hydrocarbon solvents (CHS) and organic phosphorus components was discovered. At that time the contamination had spread to a narrow plume about 1.5 kilometres downstream the site in the direction of a small creak and a local water works abstracting water from the same aquifer. It was estimated that about 24 tons of CHS and 0.5 tons of organic phosphorus components had leached to the aquifer.

The source was stopped in 1987 by remowing the waste disposal site and bring the deposits to a chemical treatment plant and in 1988 a pump-and-treat system with 4 remediation wells located along the centre line of the plume with a total abstraction of 37 m^3/hour was initiated. The abstracted water was treated in a on-site (portable) treatment plant and infiltrated into the old site in order to wash out the remaining contamination from the unsaturated zone. Under certain assumptions investigations showed that about 90% of the contamination would be extracted or degraded and that the water quality in the recipient as well as at the water works would not be threatened.

New investigations in 1991 indicated that the pollution plume had continued to migrate and had spread as far as two kilometres downstream the waste dump and approaching the recipient and the water works. The effect of the remediation programme was obviously not in accordance with the predictions and a re-evaluation of the assumptions both with respect to the hydrogeological conditions in the area and the migration and degradation characteristics of the contaminants was carried out. The findings of these investigations and the consequences on the future strategy in the area are described in the present paper.

METHODOLOGY

Geological and Hydrogeological Investigations

The geology of the Skrydstrup area is typical of the glacial plains in the western part of Denmark, just west of the Main Stationary Line of the Weischellian glacier advance in the Quaternary Period. The aquifer is unconfined and consists of glaciofluvial sand and gravel deposits. The aquifer materials are predominantly medium and coarse grained sand and gravel deposited in a braided river system on an outwash plain or in small lakes. Lenticular beds, a few metres thick, of silt and clayey silt appear discontinuously interbedded in the stratified sand and gravel deposits. The bottom of the aquifer consists of clayey till belonging to the Saale glaciation and is found at depths of 15 to 45 metres.

The estimated groundwater flow velocity is in the range of 0.5 to 0.8 m/day at a natural hydraulic gradient of 1.6 to 2.8 m/km. The sandy parts of the aquifer have a horizontal hydraulic conductivity of 1 to more than 80 m/day in contrast to the finer grained lenses with a value of less than 0.1 m/day.

Characterization of the Contamination Plume and the Contaminants

Detailed studies of the subsurface CHS contamination was carried out in the period 1987 to 1994. The investigations were designed to define the subsurface lithology, the groundwater flow directions and velocities and the extension of the contaminant plume. 39 monitoring wells and 4 abstraction wells were installed during these site investigations combined

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with an other 20 existing wells in the surrounding area. A four-nation research project aiming to develop and evaluate multilevel groundwater samplers and sampling techniques for use in standard water wells were also carried out in the area (Nilsson et al., 1995a) and contribute with valuable information. Another research project for optimization of the remedial action pumping in Skrydstrup was also carried out with the focus on the separation pumping technique (Nilsson and Jacobsen, 1993).

A CHS plume, mainly consisting of 1,1,1-trichloroethane (TCA) and tri-chloroethylene (TCE) had spread as far as two kilometres downstream of the waste disposal site very close to the recipient Jernhyt Stream (cf. Fig. 1). Plume extension studies indicated a close relation between the hydrogeology and the CHS distribution in the aquifer. It seemed to be hydraulic controlled by the morphology of the valley-like structure in the surface of the clayey till confining the bottom of the polluted aquifer.

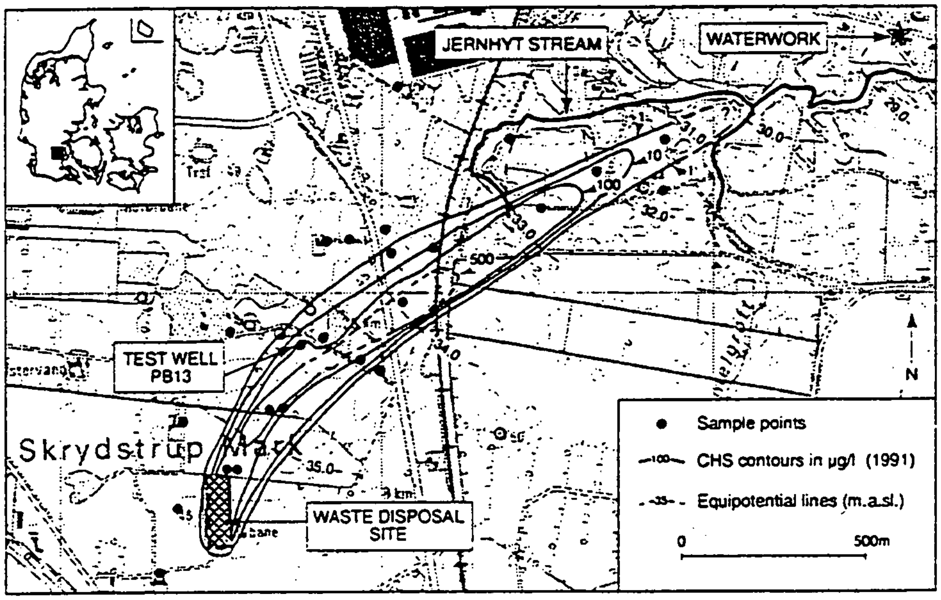

Figure 1 *Horizontal extension of the CHS plume at the Skrydstrup waste disposal site. Concentration contours represent the sum of the TCA and TCE concentrations.*

The horizons of low hydraulic conductivity probably exert control on the solute transport and spreading in the aquifer acting as hydraulic barriers. It is underlined by the fact that the contaminants are predominantly found in the upper 15 to 20 metres of the aquifer (cf. Fig. 2) in association with the medium grained sand and silt layers of lower permeability (Nilsson et al., 1995b).

Potential for Microbiological Degradation

The potential for microbial degradation of the contaminants was investigated both in sediment and pore water samples from the aquifer (Aamand and Broholm, 1991) and in a mixture of sediment and pore water samples obtained from the aerobic and anaerobic parts of the bottom sediments of Jernhyt Stream. The degradation experimennts for the bottom sediments of Jernhyt Stream were performed as described below:

Aerobic degradation experiment: 3 ml of aerobic sediment was mixed with 27 ml of pore water in a 117 ml glass bottle. A total of 7 bottles were prepared. Phosphate buffer was added to each bottle to adjust pH to 7.1. Two of the bottles were sterilized in an autoclave and used as abiotic controls. A mixture of TCA and TCE was added to each bottle from a sterile stock solution. All the bottles were closed with teflon lined stoppers and subsequently 7 ml of methane was added with a syringe. Immediately after the application of methane, the concentrations of TCA and TCE was measured in two of the bottles. The remaining 5 bottles were incubated at 15 °C. The concentrations of TCA and TCE in these bottles were measured after 20 days of incubation.

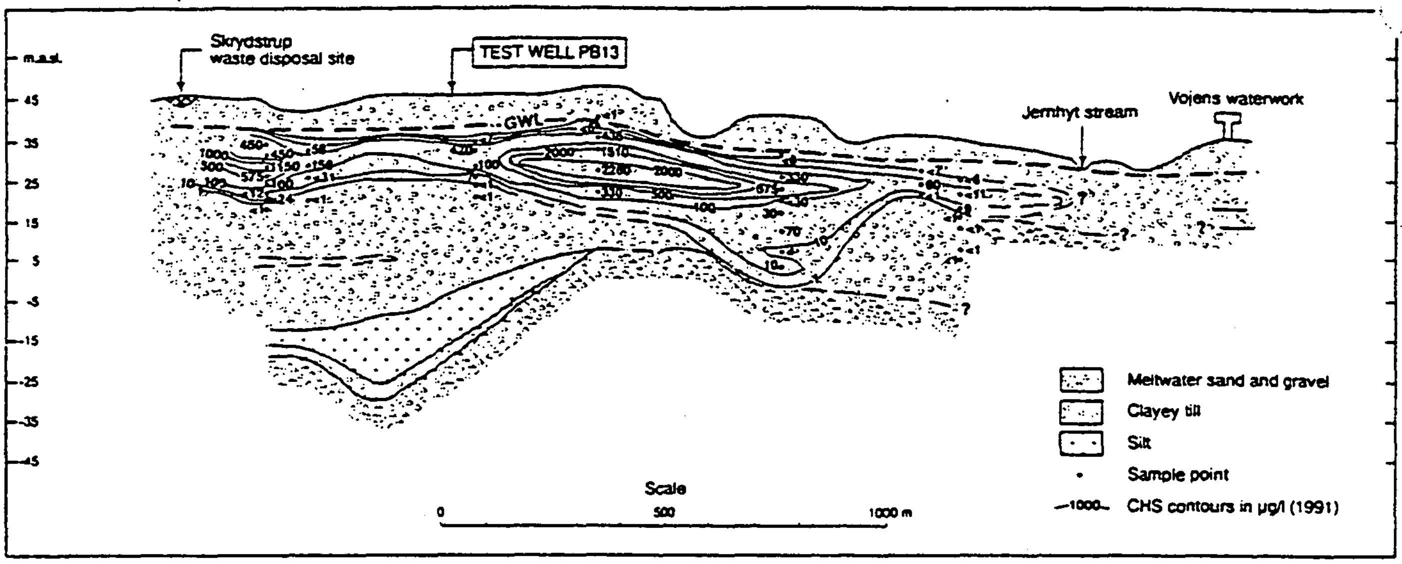

Figure 2 *Cross-sectional view of the CHS plume along the longitudinal direction of the plume. Concentration contours represent the sum of the TCA and TCE concentrations.*

Anaerobic degradation experiment: 5 ml of anaerobic sediment was mixed with 45 ml of porewater containing phosphate buffer in a 117 ml glass bottle. Oxygen had been removed from the porewater by flushing with N_2 for 2 hours. The mixture of sediment and porewater was further reduced by the addition of titanium(III) citrate. Two of the bottles were sterilized in an autoclave and used as abiotic controls. A mixture of TCA and perchloroethylene (PCE) was added to each bottle from a sterile stock solution. The bottles were closed with teflon lined stoppers. The concentrations of TCA and PCE was measured immediately in two of the bottles. The remaining 5 bottles were incubated at 15 °C and like in the aerobic experiments the concentrations of chlorinated compounds in these bottles were measured after 20 days of incubation.

Chemical analysis: TCA, TCE and PCE were extracted from the sediment/water mixture with pentane. The extract was analyzed on a gas chromatography with an electron-capture detector. The sediment/water mixtures from the anaerobic experiment were also screened by GC-MS for the presence of potential metabolites formed during anaerobic degradation of TCA and PCE (1,1-dichloroethane, 1,1 dichloroethylene and vinyl chloride).

Solute Transport Modelling

A hydrological model was constructed on the basis of the geological and hydrogeological interpretations in the area. The core of this model was the general, hydrological modelling system MIKE SHE which is an integrated tool comprising process-oriented components, each describing the major physical processes in individual parts of the hydrological cycle and in combination describing the entire hydrological cycle i.e. interception/evapotranspiration, two-dimensional overland flow and one-dimensional channel flow, one-dimensional unsaturated zone flow, three-dimensional saturated zone flow and exchange between the different components, Danish Hydraulic Institute (1993) (cf. Fig 3). A family of add-on modules for calculation of solute transport in the different components (Danish Hydraulic Institute, 1995) and advanced geo-chemical and biological reactions (Engesgaard and Hansen, 1994) ensures a great flexibility of the modelling system.

In the Skrydstrup case the hydrological model included components for the groundwater flow and solute transport as well as river flow. The three-dimensional partial differential equation for transient groundwater flow which is developed in many standard textbooks (see e.g Freeze and Cherry, 1979) is solved numerically in a finite-difference scheme using a well proven modified Gauss-Seidel implicit, iterative method (Thomas, 1973). The exchange with surface water is solved in a explicit scheme.

The advection-dispersion equation for solute transport is also derived in a number of text books (see e.g. Bear, 1972). This equation is solved in an explicit scheme called the QUICKEST method using upstream differencing for the advection term and central differencing for the dispersion term. The equations are developed to third order implying that numerical dispersion and "wiggles" are limited and this scheme is very efficient compared to other methods (Vested et al., 1992).

Pre-processing tools ensured that all available information was compiled and used in the construction of the three-dimensional hydrogeological model for the groundwater zone and the post-processing tools which included animation of the results ensured that the politicians were able to understand the conclusions of the study and take the necessary steps.

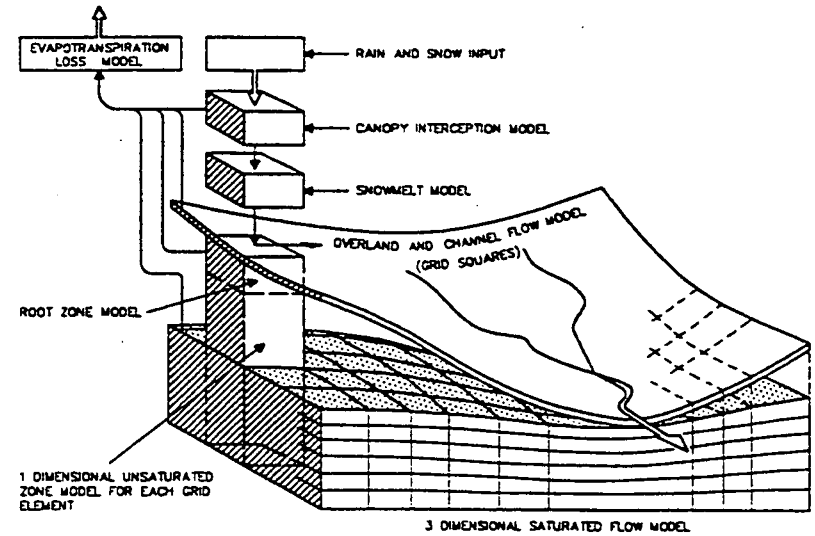

Figure 3 *Schematic presentation of the core of the MIKE SHE modelling system for simulation of water flow and solute transport in the entire land based part of the hydrological cycle.*

RESULTS

Potential for Microbiological Degradation

In the groundwater and aquifer material from the contaminated part of the aquifer no or only very little anaerobic degradation of TCA and TCE were observed (Aamand and Broholm, 1991). In the sediments from the Skrydstrup Stream the following results were obtained.

Aerobic degradation experiment: Table 1 shows the concentrations of TCA and TCE in the sediment/pore water mixture in the aerobic batch experiment before and after incubation in 20 days.

Table 1: *Removal of 1,1,1-trichloroethane (TCA) and trichloroethylene (TCE) in aerobic sediments samples from Jernhyt Stream incubated under aerobic conditions at 15 °C. The concentrations shown in the table are the means of the concentrations measured in two bottles. The concentrations marked by * are the means of three bottles.*

Compound	Day 0 μg/l	Day 20 μg/l
TCA:		
active bottle	450 ± 23	310 ± 17*
control bottle	450 ± 23	350 ± 17
TCE:		
active bottle	600 ± 93	220 ± 67*
control bottle	600 ± 93	550 ± 163

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In contrast to TCA, the removal of TCE was higher in the biologically active bottles. This observation indicates that TCE was metabolized by microorganisms present in the sediment.

It appears from the table that the removal of TCA from the bottles with active sediment and the control bottles with sterilized sediment was of the same order of magnitude. Thus, the observed removal of TCA must have been due to abiotic processes (most likely evaporation from the bottles).

Anaerobic degradation experiment:

The result from the anaerobic experiment are shown in table 2.

*Table 2: Removal of 1,1,1-trichloroethane (TCA) and perchloroethylene (PCE) in anaerobic sediments samples from Jernhyt Stream incubated under anaerobic conditions at 15 °C. The concentrations shown in the table are the means of the concentrations measured in two bottles. The concentrations marked by * are the means of three bottles.*

Compound	Day 0 µg/l	Day 20 µg/l
TCA:		
active bottle	420 ± 113	267 ± 12*
control bottle	420 ± 113	310 ± 14
PCE:		
active bottle	400 ± 28	253 ± 31*
control bottle	400 ± 28	250 ± 14

As can be seen from Table 2, the concentrations of TCA and PCE were in the same order of magnitude in the active bottles and the control bottles. This means that no significant microbial transformation of TCA and PCE occurred during the 20 day incubation period,

Solute Transport Modelling

The design of the proposed pump-and-treat system back in 1987 and the evaluation of its efficiency was based on simulations with the MOC 2D groundwater model (Konikow and Bredehoeft, 1978). This model covered an area of less than two kilometres downstream the site i.e. the recipient Jernhydr Stream was not included in the model area. The main assumptions besides the 2D modelling approach were that the retardation of the contaminants could be described by a linear Freundlich isotherm and that the degradation of the contaminants could be described by a simple first order degradation formula. The retardation factor was estimated/calibrated to a value of 4 and the half life time was estimated/calibrated to a value of 3.5 years.

Model simulations showed that with an optimal location of the remediation wells and a total abstraction rate of about 37 m^3/hour from the wells, 90% of the contamination would be extracted or degraded after 12 years. The water quality in the recipient as well as at the water works would not be threatened by the contamination since it would not reach this part of the aquifer.

A hydrological model was constructed after the new investigations of the extension of the contamination plume, the hydrogeological conditions and the potential for degradation of the contaminants. This model was based on the MIKE SHE modelling system with a three-dimensional approach for the groundwater part and addressing the interaction between the groundwater and the surface water systems. The model discretization was 50 m x 50 m in the horizontal direction and 2 m in the vertical direction.

The flow part of the model was calibrated (adjusting hydraulic conductivities of the identified geological units in the area) against water table maps and (base-) flow in the Skrydstrup Stream as a stationary model. The stationary approach was justified by the fact that only very little changes in groundwater flow directions as well as in base-flow had been observed in the 5 to 7 years of observation material which was available. A mean transmissivity of 30 x 10^{-3} m^2/s was found for the main part of the aquifer.

As a result of the latest findings the solute transport part of the model only included retardation and no degradation since the solute main pollutants is TCA and there are aerobic conditions in the aquifer. Very detailed source characterization based on annual air photos of the area combined with information from the factory that had deposited the CHS barrels alloved to establish a reliable description of the transient source concentration. Assuming that the source concentration was correct the model was calibrated (adjusting porosity and dispersivities) against the location of the contamination plume and the concentration of the water extracted during the remediation period. Rather good agreement was achieved with a mean porosity equal to 0.2, a longitudinal dispersivity equal to 2 m and both the horizontal and the vertical transverse dispersivities equal to 0 m. Due to local heterogeneities some divergence in concentration level occurred in some of the investigation wells.

Model simulations showed that 95% of the groundwater contaminated with infiltrating contaminants at the waste disposal site recharges to the Jernhyt Stream at its upper 1.8 km. Furthermore, the results indicated that the contamination plume had already reached a part of the stream and that contaminated groundwater was recharging to the stream (this was later demonstrated by drilling wells close to the stream). The model simulations on the effectiveness of the pump-and-treat system (cf. Fig. 4) gave the following main results:

* about 2.5 tons of the 24 tons of CHS have been removed during the first five years of operation of the present pump-and-treat arrangement. Only about 1 ton will be extracted during the next five years and the efficiency will continue to drop due to the propagation of the plume;

* pumping from additional and/or other wells will increase the amount of contamination that can be extracted, but even with unrealistic high (due to the capacity of the on-site treatment plant) pumping rates of about 110 m^3/hour from optimally located wells it is not possible neither to recover more than 50% of the present contamination in the aquifer nor to prevent a considerable amount of contamination to recharge to the recipient Jernhyt Stream;

* after 25 years of pumping a considerable amount of contamination will still remain in the aquifer;

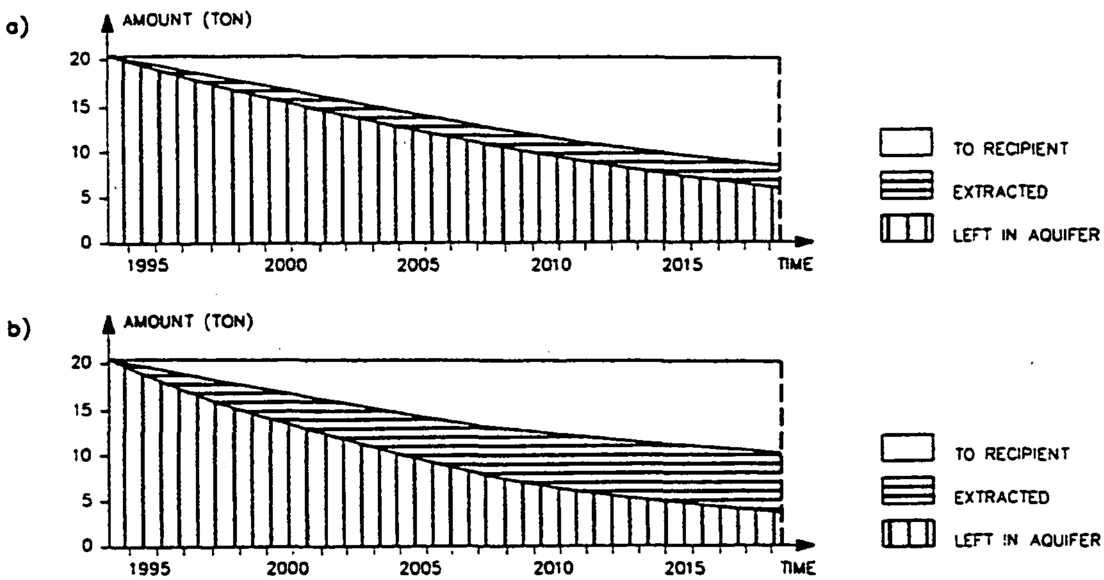

Figure 4 *Calculated distribution between contamination extracted by pumping, recharging to Jernhyt Stream and remaining in the aquifer the next 25 years for two different pumping strategies; a) continuation of the "present" situation (4 wells, total 37 m^3/hour) and b) pumping from 6 optimally located wells with a total pumping rate of 110 m^3/hour.*

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With these findings in mind it was decided to stop the present pump-and-treat arrangement and concentrate the future work on measuring the propagation of the plume. Special focus will be given to the riparian areas along the recipient and the recipient it self.

DISCUSSION AND CONCLUSIONS

Although much uncertainty and limitations are introduced applying a mathematical, hydrological modelling system in the design and investigations of remedial measures as pump-and-treat systems they are useful tools for compiling all available information in a consistent manner since the application and calibration of a model provides a method for successive confrontation of hypotheses and interpretations of the system and available measurements. It seems that mathematical modelling is the only way to effectively investigate and optimize location and pumping rates for remediation wells in a pump-and-treat system especially when contaminants are "reactive" and interacts with both aquifer material and other species present in the groundwater. In this connection it is of great importance to investigate the degradation of the contaminants under "natural" conditions and include these findings in the transport modelling.

The contamination downstream Skrydstrup waste disposal site mostly consisted of TCE, TCA and PCE. Most chlorinated solvents can only be degraded by bacteria in the presence of other, easily degradable substrates that will serve as the primary source of energy for the bacteria (Vogel, 1987). Under aerobic conditions methane has been shown to be one of the primary substrates that will easily induce microbial degradation of chlorinated solvents (Vogel, 1987 and Broholm, 1991). Under anaerobic conditions a variety of substrates produced during the anaerobic fermentation of organic matter can serve as primary substrates and induce a transformation of the chlorinated compounds.

In Jernhyt Stream methane oxidizing bacteria capable of metabolizing TCE but not TCA were apparently present in the upper sediments where methane produced in the deeper parts diffuses into an aerobic environment. In the anaerobic sediments, no degradation of TCA and PCE was observed during a 20 days incubation period.

Since TCA accounts for most of the groundwater pollution, no degradation was assumed in the solute transport model applied to assess the efficiency of the present and proposed pump-and-treat systems. The conclusion from the modelling work was that remedial measures including pump-and-treat systems will have limited effect on the propagation of the pollution plume as most of the contaminants will recharge to the stream without a significant removal by biological processes.

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Chapter 23

MIKE SHE

J. C. Refsgaard and B. Storm

23.1. INTRODUCTION

MIKE SHE is a comprehensive deterministic, distributed and physically-based modelling system for the simulation of all major hydrological processes occurring in the land phase of the hydrological cycle. It simulates water flow, water quality and sediment transport.

MIKE SHE is a further development based on the SHE modelling concept developed by a European consortium of three organizations: the Institute of Hydrology (UK), the French consulting firm SOGREAH and DHI (Abbott et. al. 1986).

MIKE SHE is a fourth-generation, user-friendly modelling package comprising a number of comprehensive pre- and post-processors including digitizing, graphical editing, contouring, grid-averaging and graphical-result presentation with options for display of animations. It has been designed for efficient application on relatively low-cost, high-performance workstations.

MIKE SHE is applicable to a wide range of water resources and environmental problems related to surface water and groundwater systems and the dynamic interaction between these. Typical areas of application are:

- river basin planning
- water supply
- irrigation and drainage
- contamination from waste disposal sites
- impacts of farming practises (including the use of agrochemicals and fertilisers)
- soil and water management
- effects of changes in land use

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- effects of changes in climate
- ecological evaluations, including those associated with wetland areas.

MIKE SHE is applicable on spatial scales ranging from a single soil profile (for infiltration studies) to large regions, which mayinclude several river catchments. It has been tested and proved in a large number of research and consultancy projects. The experience record covers a wide range of climatological and hydrological regimes.

23.2. INTEGRATED MODULAR STRUCTURE

The basic MIKE SHE Module is MIKE SHE WM for the description of water movement in the area under study. The WM Module itself has a modular structure with one component dedicated to each hydrological process. Several components include alternative options for describing certain specific processes.

The user can produce his own model configuration adapted specifically to the local hydrological conditions and the purpose of the study.

The following add-on modules are available for water quality, soil erosion and irrigation studies:

- **MIKE SHE AD** - advection and dispersion of solutes
- **MIKE SHE GC** - geochemical processes
- **MIKE SHE CN** - crop growth and nitrogen processes in the root zone
- **MIKE SHE SE** - soil erosion
- **MIKE SHE DP** - dual porosity
- **MIKE SHE IR** - irrigation

In this presentation only the WM module is described in details, while a brief introduction is given to the AD module.

23.3. WATER MOVEMENT MODULE (MIKE SHE WM)

23.3.1. Programme Structure

The overall model structure is illustrated in Fig. 23.1. MIKE SHE WM has been designed with a modular programme structure comprising six process-

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oriented components, each describing the major physical processes in individual parts of the hydrological cycle and in combination describing the entire hydrological cycle:

- Interception/evapotranspiration (ET)
- Overland- and Channel Flow (OC)
- Unsaturated Zone (UZ)
- Saturated Zone (SZ)
- Snow Melt (SM)
- Exchange between Aquifer and Rivers (EX)

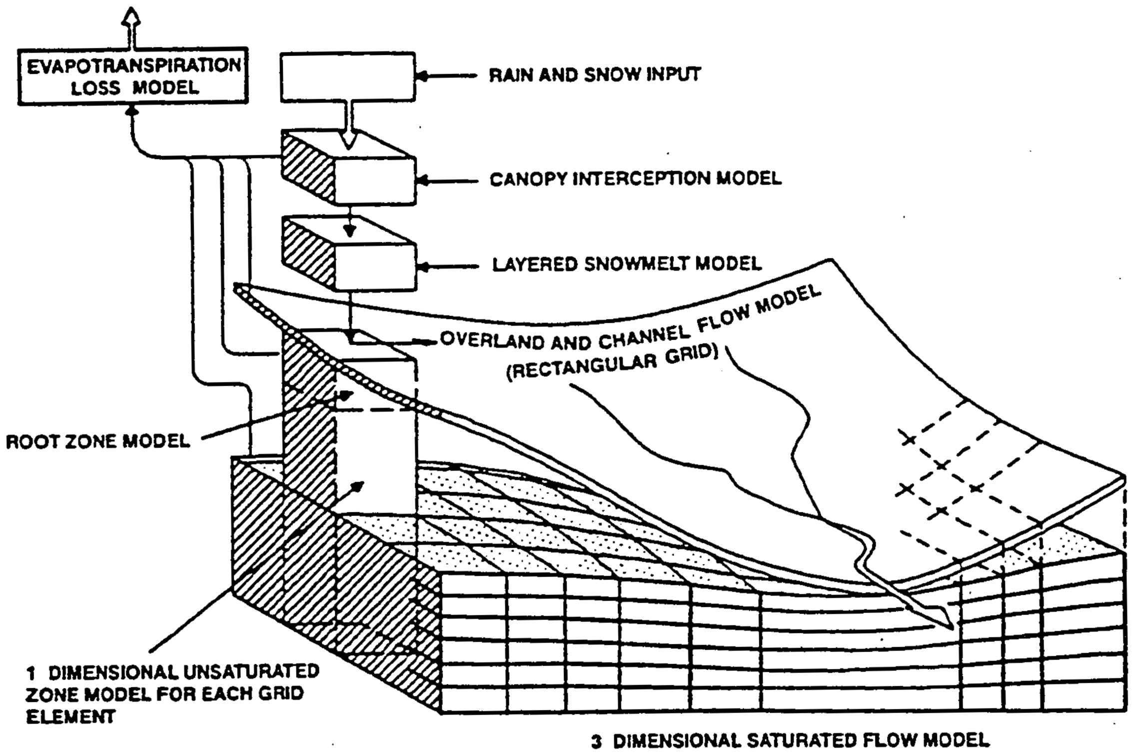

Fig. 23.1. Schematic representation of the components of the MIKE SHE.

The modular form of system structure, or architecture, ensures a great flexibility in the description of the individual physical processes. Data availability or specific hydrological conditions may favour one model description compared to another. By ensuring that the data flow between components are unchanged, alternative methods which are generally accepted in a certain geographical region or a country can be included in the MIKE SHE WM system if required.

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The governing partial differential equations for the flow processes are solved numerically by efficient and stable finite difference methods in separate process components. All process descriptions operate at time steps consistent with their own most appropriate temporal scales. Hence the processes may be simulated using different time steps which may be updated during the simulation and coupled with the adjoining processes as and when their time steps coincide. The facility allows for a very efficient operation, making it possible to carry out simulations extending over long periods of time.

Individual components can also be operated separately to describe a single process. This may be relevant in a range of applications, where only rough estimates of data exchange from other parts of the hydrological cycle are required. An example could be a groundwater study where only approximate recharge estimates may be required and a full coupling to the unsaturated zone above the groundwater table is unimportant.

The ability to provide an integrated description of the various processes, despite different time scales, is the most important feature of MIKE SHE WM. This integration has probably been the largest problem encountered during its development and provides a unique feature. Perhaps the most difficult coupling is the one between the unsaturated zone and the groundwater components, which is described in Storm (1991).

23.3.2. Frame Component

A FRAME component coordinates the parallel running of the process components by selecting their different time scales and organizing their data interchanges. Its primary functions include:

(1) Controlling the sequence in which each component is called to perform its computations. For some components there may be different time steps as well as changes in the time steps during a simulation, depending on the rate of hydrological response. If the time steps differ between components, it is necessary to accumulate calculated values over a period before they are transferred to another. In general, the larger time steps are used in the saturated zone component compared to the other components.

(2) Controlling which components are included in the simulation. Dummy versions, which simply provide the necessary boundary variables for the other components may substitute process components if the concerned processes are irrelevant for the application.

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(3) Controlling which option of each process description is used if more than one method is available for a component.

(4) Controlling the exchange of data between components. The results provided by one component will in some cases need to be processed into a different form for input to another component.

(5) Controlling the output variables to be stored on disk for postprocessing and the time interval with which each variable is stored.

The data flow between components is illustrated in Fig. 23.2.

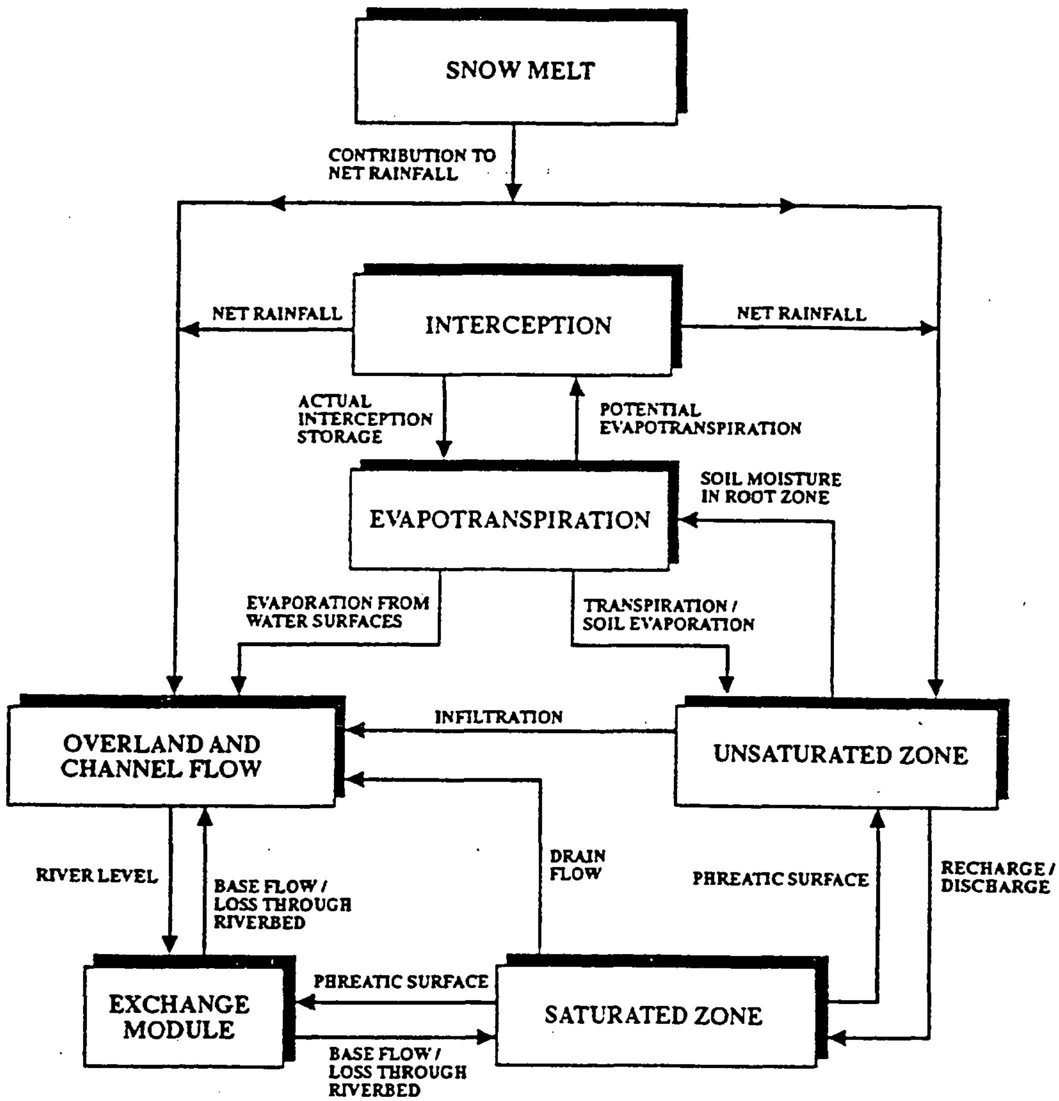

Fig. 23.2. Data flow between the components of the MIKE SHE WM.

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23.3.3. Interception and Evapotranspiration Component

During rainfall, part of this will be intercepted by the vegetation and subsequently lost by evaporation.

It is assumed that the interception is either depending on the rainfall rate and/or the interception capacity. The importance of interception depends very much on vegetation type, development stage, density of vegetation and the climatic conditions. Dense forest canopies may account for a considerable interception loss, whereas for shorter and sparser vegetation, such as grass and agricultural crops, the evaporation loss may be much smaller and often insignificant.

Evapotranspiration involves the transfer of large quantities of water. In temperate areas approximately 70% of the annual precipitation is returned to the atmosphere, while under arid conditions it almost equals the rainfall. For this reason the prediction of the actual evapotranspiration plays a key role in many water resources studies.

The evapotranspiration is the total process of evaporation from soil and water surfaces and transpiration, the water uptake by plant roots that is transpired from the leafy parts of the plant. The spatial and temporal variation in the evapotranspiration rate in a catchment depends on multiple factors such as water availability in the rootzone, the aerodynamic transport conditions, plant physiological factors etc.

At present, two alternative formulations of the interception/ evapotranspiration process are available in MIKE SHE WM.

(a) The Rutter Model/Penman-Monteith Equation

The interception is modelled by a modified Rutter model (Rutter et. al., 1978). This calculates the evaporation, the actual storage on the canopy, and the net rainfall reaching the ground surface as canopy drainage and throughfall.

The actual evapotranspiration rates are calculated by the Penman-Monteith equation using canopy resistances. The potential evapotranspiration is calculated directly using climatological and vegetation data.

(b) The Kristensen-Jensen Model

The interception storage is calculated based on actual leaf area index and an interception capacity coefficient. The net rainfall is calculated by a simple water balance approach.

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The actual evapotranspiration is calculated on the basis of potential rates and the actual soil moisture status in the root zone (Kristensen and Jensen, 1975).

EQUATIONS

(a) The Rutter model/Penman-Monteith equation

Interception

The Rutter model is essentially an accounting procedure for the amount of water stored on the canopy. The rate of change in storage is given by

$$\frac{\partial C}{\partial t} = Q - Ke^{b(C-S)} \tag{23.1}$$

where

$$Q = \begin{cases} P_1 \, P_2 \left(P - E_p \, C / S\right) & when \; C \leq S \\ P_1 \, P_2 \left(P - E_p\right) & when \; C \leq S \end{cases} \tag{23.2}$$

with the notation as given at the end of this section.

Evapotranspiration

The Penman-Monteith equation (Monteith, 1965) for predicting the actual evapotranspiration rates is

$$E_a = \frac{\Delta \, R_n + \dfrac{\rho \, C_p \, \delta_e}{r_a}}{\lambda \left\{\Delta + \gamma \left(1 + r_s / r_a\right)\right\}} \tag{23.3}$$

The total actual evapotranspiration calculated for each grid square depends on the wetness of the canopy and the degree of ground covered by the canopy:

$$E_t = P_1 \, P_2 \, E_p \, C / S + E_a \left(1 - C / S\right) P_1 \, P_2 + E_s \left(1 - P_1 \, P_2\right) \tag{23.4}$$

(b) The Kristensen-Jensen model

Interception

The intercepted water storage capacity is calculated by:

$$S = S_{int} LAI \tag{23.5}$$

The evaporative demand constraint is first applied to the intercepted water and if this is not satisfied the remaining part is applied for transpiration and soil evaporation. If the interception storage capacity is exceeded during rainfall, the surplus of rain will be calculated as throughfall.

Evapotranspiration

The evaporative demand is only met if the soil moisture content in the rootzone is sufficient. For lower moisture contents the actual transpiration is calculated according to a procedure illustrated in Fig. 23.3.

The soil evaporation demand is reduced below the potential value for moisture contents below field capacity, according to the procedure illustrated in Fig. 23.4.

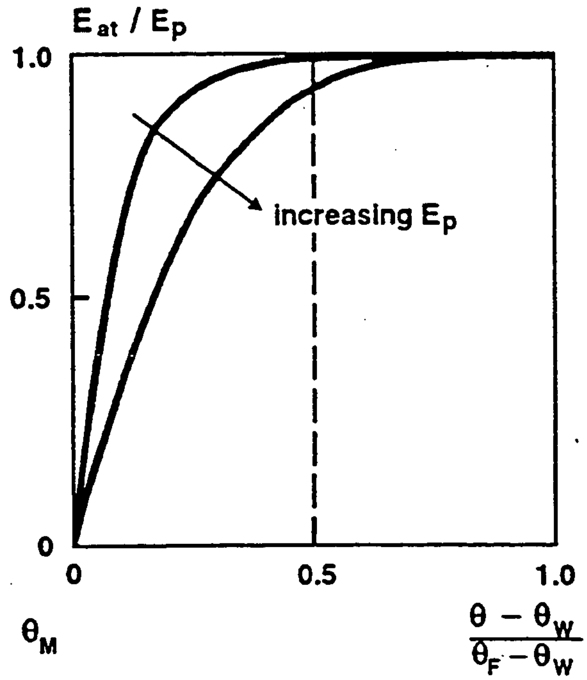

Fig. 23.3. Relationship between actual transpiration and soil moisture content in the Kristensen-Jensen model.

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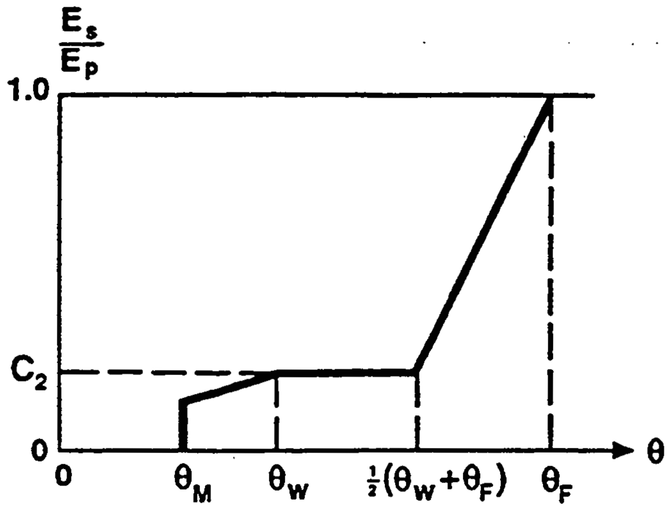

Fig. 23.4. Relationship between actual soil evaporation and the and soil moisture content in the Kristensen-Jensen model.

Symbol List

E_t - total evapotranspiration rate

E_a - actual evapotranspiration

E_{at} - actual transpiration

E_p - potential evapotranspiration

E_s - soil evaporation

C - actual water depth on canopy (mm)

S_{int} - interception storage coefficient (mm)

S - canopy storage capacity (mm)

P - rainfall rate (mm/s)

P_1 - proportion of ground planview hidden by vegetation

P_2 - ratio of total leaf area to area of ground covered by vegetation

K, b - canopy drainage parameters

t - time (s)

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R_a - net radiation minus energy flux into the ground (W/m^2)

Δ_e - slope on specific humidity - temperature curve (mb/C°)

ρ - density of air (Kg/m^3)

C_p - specific heat capacity of air (J/kg/C°)

δ_e - vapour pressure deficit (mb)

r_a - aerodynamic resistance (s/m)

r_s - canopy resistance (s/m)

λ - latent heat of vaporization (J/kg)

γ - psychometric constant (mb/C°)

LAI - leaf area index

θ_F - soil moisture content at field capacity

θ_W - soil moisture content at wilting point

θ_M - irreduable soil moisture content

Solution Technique

The Rutter model is solved by analytical integration. All other equations are solved directly.

The transpiration is calculated as sinks at each computational node in the rootzone and the total transpiration is found as a weighted average according to the actual root mass distribution. For the top node, the additional loss due to soil evaporation is also considered.

Examples of Output

The ET component calculates, at each grid square where calculation are performed with the UZ component, time series of e.g. evaporation from interception storage and ponded water, soil evaporation, and transpiration. This is illustrated in Fig. 23.5.

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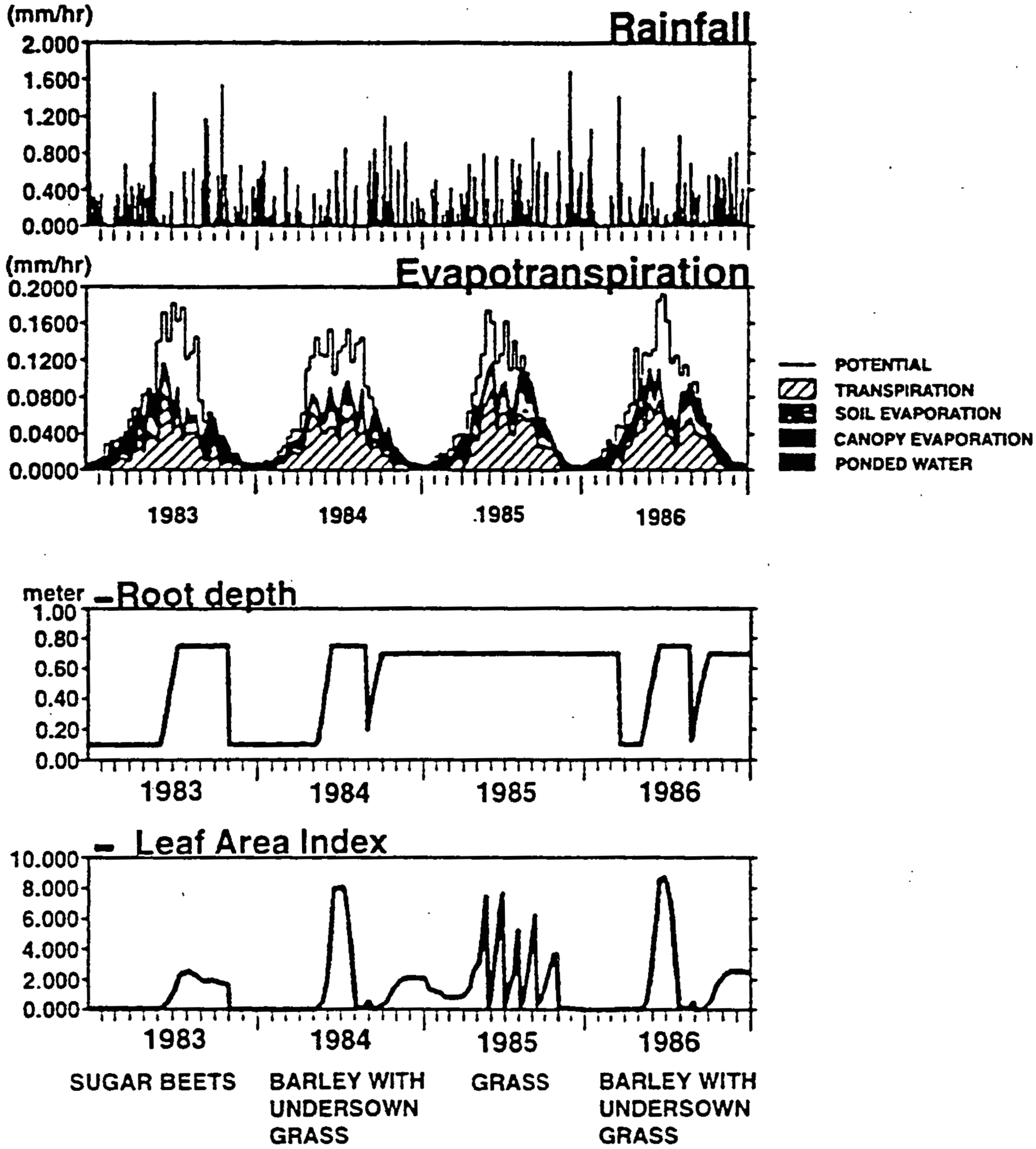

Fig. 23.5. Example of output from ET calculations for a four year period in a single grid square.

23.3.4. Overland - and Channel Flow Component

Water accumulated on the soil surface during heavy rainfall or when the groundwater table rises to the ground surface responds to gravity by flowing down-gradient over the land surface enroute to the stream channel system. From here it discharges through the channels (rivers) to the outlet of the catchment.

During its journey to the streams, the flowing water may diminish because of evaporation or infiltrate in areas with more permeable soils.

The stream channel system is assumed to run along the boundaries of the grid squares.

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EQUATIONS

The overland flow process is described by solving the equations of continuity and conservation of momentum in two horizontal directions (the Saint Venant equations). In the latter the diffusive wave approximation is applied.

$$\frac{\partial h}{\partial t}+\frac{\partial(uh)}{\partial x_i}+\frac{\partial(vh)}{\partial x_j}=q$$

$$\frac{\partial h}{\partial x_i}=S_{ox_i}-S_{fx_i} \qquad x_i - direction \tag{23.6}$$

$$\frac{\partial h}{\partial x_j}=S_{ox_j}-S_{fx_j} \qquad x_j - direction$$

The Strickler/Manning-type law for each friction slope is used giving the following relations between velocities and flow depths.

$$uh = K_{x_i}\, I_{x_i}^{1/2}\, h^{5/3}$$

$$vh = K_{x_j}\, I_{x_j}^{1/2}\, h^{5/3} \tag{23.7}$$

The channel flow is calculated by an equivalent set of equations, but in one dimension only.

$$\frac{\partial A}{\partial t}+\frac{\partial(Au)}{\partial x}=Q$$

and (23.8)

$$\frac{\partial h}{\partial x}=S_{ox}-S_{fx}$$

Symbol list

$h(x_i, x_j)$	-	local water depth exceeding any detention storage (m)
x_i, x_j	-	space coordinates (m)
t	-	time (s)
$u, v(x_i, x_j, t)$	-	flow velocities (m/s)
$q(x_i, x_j, t)$	-	source/sink per unit horizontal area ($m^3/s/m^2$)

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$S_{oi}, S_{oj}(x_i, x_j)$	-	bed slope in x- and y-directions
$S_{fi}, S_{fj}(x_i, x_j)$	-	friction slope in x- and y-directions
$K_i, K_j(x_i, x_j)$	-	Strickler roughness coefficients in x- and y-directions ($m^{1/3}/s$)
$I_i, I_j(x_i, x_j)$	-	gradients of the water surface levels in x- and y directions
$A(x)$	-	cross-sectional area of the channel (m^2)
$Q(x)$	-	source/sink (lateral in- and outflow from overland flow, drainage flow and exchange with the aquifer) (m/s)

Important Parameters

The Strickler roughness coefficients used both in the overland flow and the river flow descriptions influence the timing and shape of the simulated hydrograph simulations. In particular for the overland flow part these parameters may be subject to calibration.

Solution Technique

Implicit finite difference techniques are used for both the overland flow (Thomas, 1973) and the channel flow (Preissman and Zaoui, 1979). The former uses a modified Gauss-Seidel iterative solution scheme, analogous to that often used for the groundwater flow.

Examples of Output from the OC Component

The OC component calculates the temporal and spatial ponding depths and flows on the ground surface. In the river system the temporal and spatial variations in water levels and discharge are produced. A snapshot from a simulation is shown in Fig. 23.6. The figure illustrates the ponded water on the land surface in a fence diagram of the water level in the channel system.

23.3.5. Unsaturated Zone Component

The unsaturated zone is a crucial part of the hydrological system in a catchment. It plays an important role in many modelling applications, e.g. for recharge estimation, surface-groundwater interaction and agricultural pollution. The unsaturated zone refers here to the mostly-unsaturated soil

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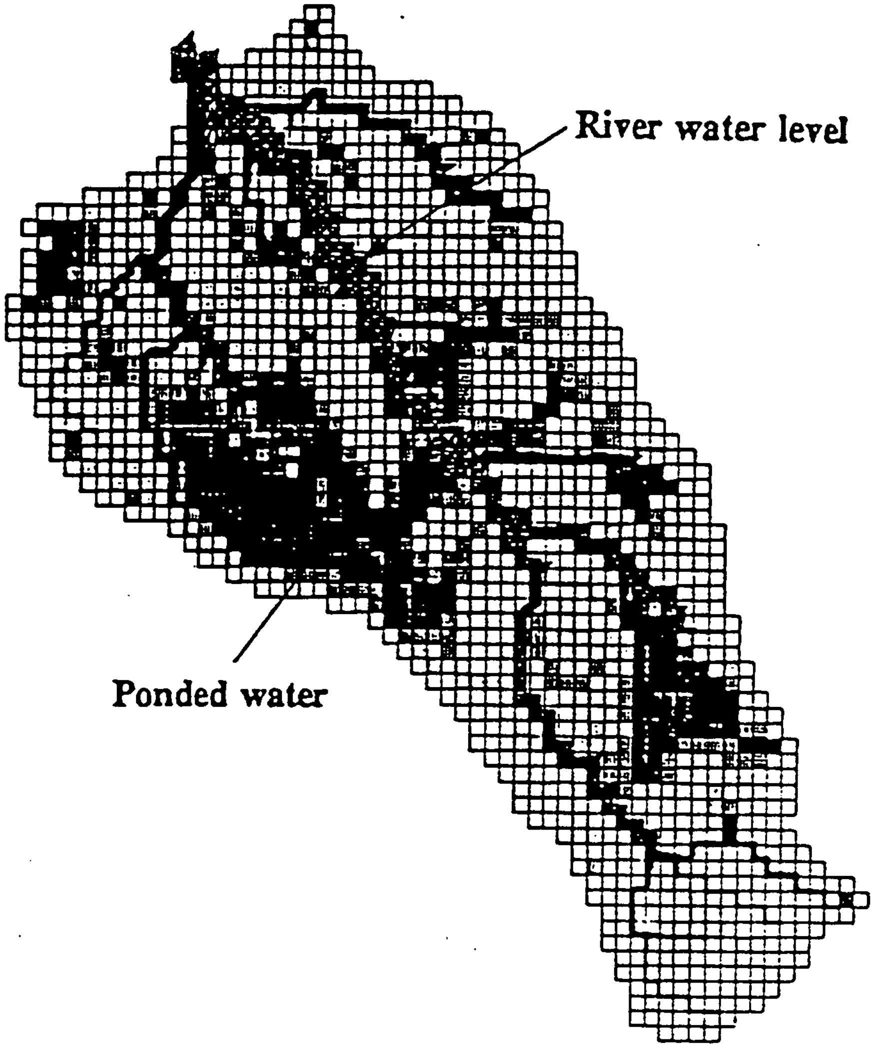

Fig. 23.6. Example of output from the OC calculations. Spatial variation of depths of water levels in river and on surface at a given time.

profile extending from the land surface down to the groundwater table. The profile is usually heterogeneous, consisting of horizons with distinct differences in the physical properties of the soil.

The unsaturated zone is characterized by cyclic fluctuations in the soil moisture as water is removed from the soil profile by evapotranspiration and percolation and replenished by rainfall.

Percolation may introduce a rise in the water table, whereas upward flow from the groundwater due to capillary rise may occur in areas with a high groundwater table and high evaporation demands.

Unsaturated flow can usually be considered vertical since gravity plays a major role during the percolation of water. The unsaturated flow is therefore only represented in MIKE SHE WM by a vertical flow component, which fulfils the requirements of most situations. However, this assumption may limit the validity of the flow description in some special cases, e.g. on very steep hillslopes with contrasting soil properties in the profile.

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EQUATIONS

In its most comprehensive mode, MIKE SHE WM solves Richards equation for one-dimensional vertical flow, which includes the effects of:

- gravity
- soil suction
- soil evaporation
- transpiration

in the form:

$$C\frac{\partial\psi}{\partial t}=\frac{\partial}{\partial z}\left(K\frac{\partial\psi}{\partial z}\right)+\frac{\partial K}{\partial z}-S \tag{23.9}$$

The equation is solved over all representative grid squares in the model area.

In many soils pronounced macropore flow is observed. In addition to Richards equation, which only considers the water flow in the micropores (soil matrix), an empirical bypass function is introduced, which calculates a direct instantaneous groundwater recharge as a certain percentage of the net rainfall and the actual soil moisture conditions in the root zone.

Symbol list

$\psi(z,t)$	-	pressure head (m)
t	-	time (s)
z	-	vertical space coordinate (m)
C	-	soil water capacity (m^{-1})
$K(\theta,z)$	-	hydraulic conductivity (m/s)
θ	-	soil moisture content
$S(z,t)$	-	source /sink term (e.g. root extraction) (s^{-1})

Important Parameters

Two non-linear functional relationships are required to solve Richards' equation: the unsaturated hydraulic conductivity function $K(\theta)$ and the soil moisture retention curve, describing the $\theta-\psi$ relation. Both functions should be known for any soil type included in any soil profile within the model area.

In applications made on the catchment scale, these functions are not known in detail. Usually representative values can be obtained for the soil types. The parameters may therefore be subject to calibration.

Solution Technique

Richards' equation is solved numerically by an implicit finite difference technique using the double-sweep algorithm. The variables are defined at every computational node in the vertical.

The time-varying upper boundary at the land surface can shift between a flux-controlled boundary (net rainfall) and a soil-controlled head boundary during ponding. The time-varying lower boundary is usually the groundwater table, specified by a positive value of ψ at the computational node just below the groundwater table. If unsaturated conditions develop in the entire soil profile a zero-flux boundary is used at the impermeable interface until saturated conditions build up from the bottom.

A coupling procedure between the unsaturated and the saturated solutions is included to compute the correct soil moisture and the water table dynamics in the lower part of the soil profile. This overcomes any problems arising from the parallel running of the individual components.

Example of Output

The UZ component computes the temporal variation in soil moisture in all nodes above the groundwater table. The infiltration and recharge rate is also calculated. The temporal variation in soil moisture for two distances to the groundwater table is illustrated in Fig. 23.7.

23.3.6. Groundwater Component

Saturated subsurface flow plays a significant role in the hydrological cycle. During drought periods it provides and sustains streamflow through baseflow, while during storm events it may contribute significantly to the stormflow as well as influence the magnitude of overland flow provided by the rising water table.

Aquifers in catchments can contain large water resources storage which, besides natural changes, may be subject to abstraction for water supply and irrigation. The influence of this human activity may influence the natural recharge and discharge properties and thereby change the flow regime in the catchment.

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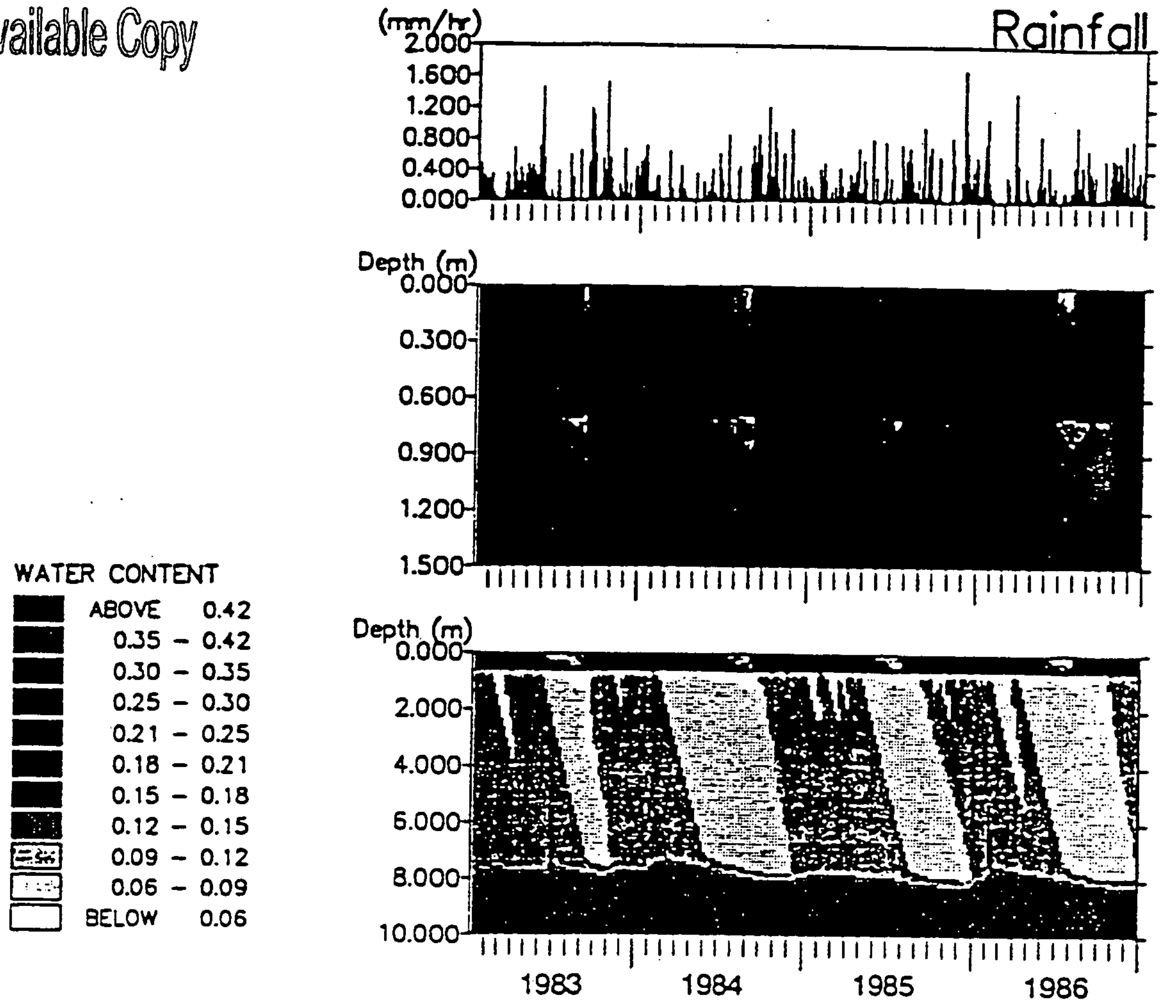

Fig. 23.7. Example of output from the UZ calculations. Temporal variation of soil moisture content at two different UZ-columns.

EQUATIONS

The temporal and spatial variations in the hydraulic head are calculated by solving the partial differential equation of groundwater flow in three dimensions:

$$\frac{\partial}{\partial x_i}\left(K_{ij}\frac{\partial h}{\partial x_j}\right) = S_s\frac{\partial h}{\partial t} + R \quad i,j = 1,2,3 \qquad (23.10)$$

The equation is applicable both for a single-layered aquifer as well as a multi-layered aquifer system.

For applications on single aquifer systems, where a two-dimensional flow solution is adequate, the equation may degenerate to two dimensions by integrating over the saturated depth.

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Symbol list

$h(x_i)$	-	hydraulic head (m)
x_i	-	space coordinate (m)
$K(x_{ij})$	-	hydraulic conductivity tensor (m/s)
$S_s(x_{ij})$	-	specific storage
$R(x_{ij})$	-	volumetric flow rate via sources or sinks per unit volume ($m^3/s/m^2$)

The source/sink term R includes

- flow exchange with the unsaturated zone
- river exchange rates
- abstraction/injection rates
- evaporation losses
- flow through drain pipes

Important Parameters

A MIKE SHE WM generated model which includes the groundwater component will require information about the hydrogeology in terms of the hydraulic conductivity K (in vertical and horizontal directions), and the storage coefficient for unconfined and confined conditions. Based on a hydrogeological model for the aquifer system, parameter values for each computational node is generated by interpolation with the pre-processor.

Solution Technique

The equation is solved numerically by a finite-difference method using a modified Gauss-Seidel implicit, iterative scheme (Thomas,1973) providing a value for the hydraulic head in time at each computational node. The discretization may either be a rigid network for fully three-dimensional flow or following the geological layering for a quasi-three dimensional flow as illustrated below. In the latter case, each layer (aquifer and aquitard) is represented by one computational node only, and the vertical variations in the groundwater head within each layer is neglected.

Examples of models for a single unconfined aquifer (Karup Å) and a multilayer aquifer system (Langvad Å) respectively, are shown in Fig. 23.8.

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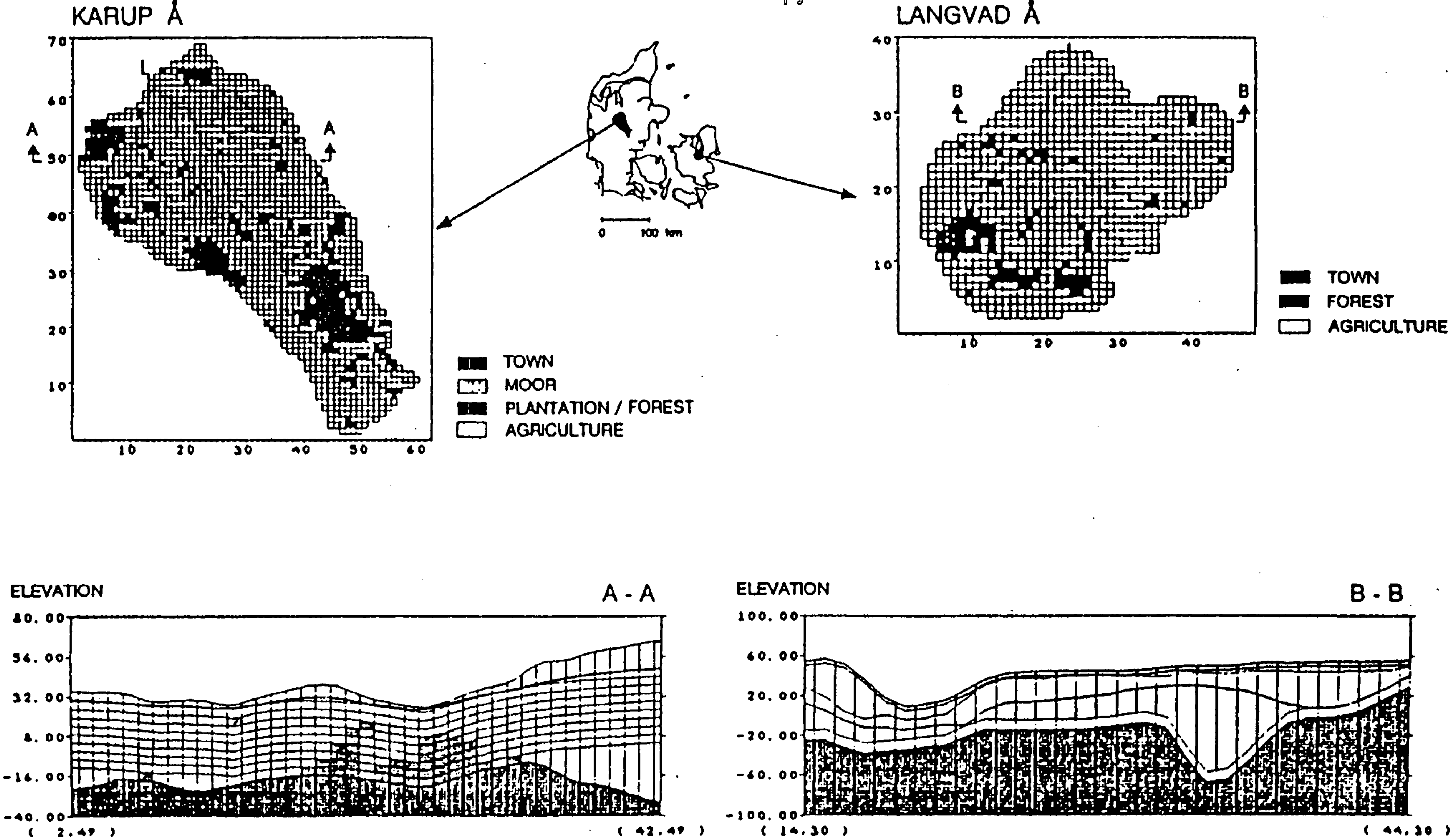

Fig. 23.8. Example of model setup for the SZ component for an unconfined aquifer (a. Karup Å) and multilayer aquifer system (b. Langvad Å).

The main characteristics of the SZ component and the type of regimes for which it is valid for are described in Table 23.1.

TABLE 23.1.

The main characteristics of the SZ component of the MIKE SHE WM

Soil/Rock Characteristics	**Fluid Conditions**
Confined aquifers	Single fluid
Aquitards	
Watertable aquifers	
Multiple aquifer	**Flow Conditions**
Heterogeneous	
Anisotropic	Unsaturated *
Single soil layer	Transient
Layered soils	
	Boundary Conditions
Geometry	*Internal grid squares*:
	Time-varying point source
1-dimensional	Time-varying line source
- horizontal	Time-varying distributed limited area (infiltration to top-layer)
- vertical	
2-dimensional	*At the model boundary*:
- horizontal	Specified time-varying distributed head
- vertical	Specified constant flux
3-dimensional	Specified constant head gradient

* includes the unsaturated zone component

Examples of Outputs

The SZ component provides information on the hydraulic heads and flows in time and space. An example of different maps to illustrate simulated hydraulic heads is shown in Fig. 23.9.

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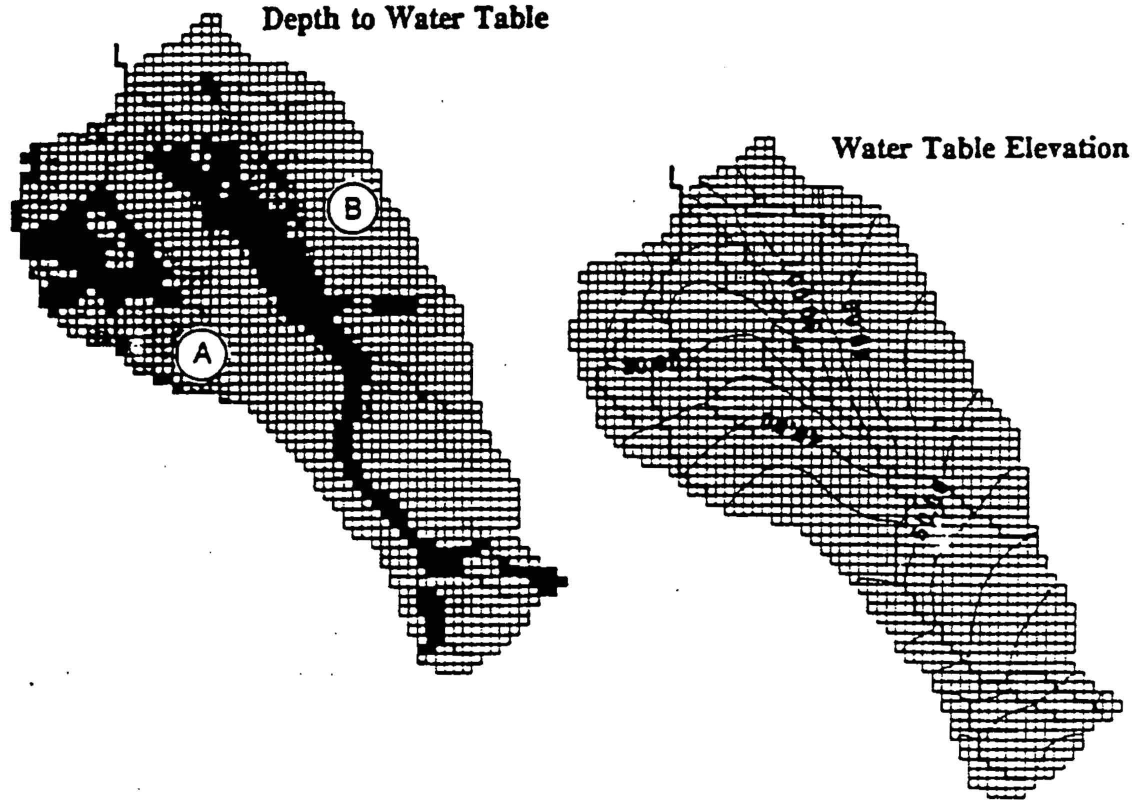
Spatial Distribution:



Time series:

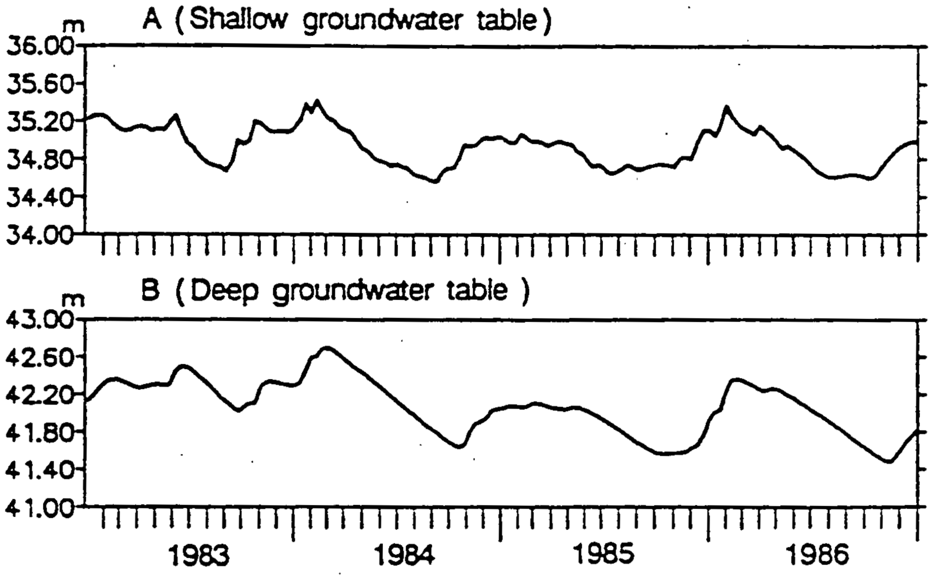


Fig. 23.9. Example of output from the SZ component illustrating both the temporal and spatial variation of ground water tables for an unconfined aquifer.

23.3.7. Aquifer-River Exchange Component

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The river system usually influences a large part of the groundwater system at its traverse of the catchment in many directions. The river will control the groundwater head in both the horizontal and vertical directions, defining the recharge and discharge areas.

Since the river surface area is often small compared to the total catchment area, it can in many applications be represented in a separate node system running along the boundaries of the grid squares acting as a line source/sink.

The interaction between river and groundwater is assumed to take place in the middle of the intermediate river links connecting adjacent river computational nodes at the corners of the grid squares.

EQUATIONS

The EX component includes two options for describing the river/groundwater exchange:

(1) The river is in full contact with the groundwater aquifer (but not necessarily fully penetrating the aquifer)

(2) A riverbed lining of low permeability separates the river from the groundwater aquifer.

In both options, Darcy's law is applied taking the additional head loss around the river bottom approximately into account.

In option (1), assuming the degree of river penetration is the limiting factor for the flow exchange, only the low area between the groundwater table and the river bed is taken into account. In option (2) the flow exchange is calculated as two resistances in series, one in the aquifer and one situated across the riverbed lining.

23.3.8. Snow Melt Component

The SM component includes two options:

(1) An energy balance approach accounting for the energy and mass flux and for changes in the structure of the snow pack (Morris, 1982)

(2) A simple degree-day approach

23.3.9. Parameters and Data Requirements

The application of distributed, physically-based models such as MIKE SHE requires the provision of large amounts of parametric and input data. Some of these may be time-dependent.

It is important to notice that MIKE SHE allows the user to utilize a large quantity of data, but it does not necessarily restrict the use of MIKE SHE if not all data are available. A model that is set up, or "instantiated", using

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MIKE SHE can be simplified according to its users conceptualization of the natural system and the data availability. Although MIKE SHE is an advanced modelling system, it is easy to apply and provides a greater flexibility as compared to many simpler models when describing natural systems.

No matter which mathematical model approach is used, the user should emphasize the reliability of the model output in view of the limiting factors for describing the natural system.

It is important to recognize that MIKE SHE WM requires calibration if it is applied on a different scale from that for which the equations have been developed and the parametric data are representative. In this connection it is important to realize the representativeness of model output compared to measured values.

Table (23.2) describes the parametric and input data for MIKE SHE WM.

TABLE 23.2.

Data and parameter requirements for each grid square or channel link

Frame	
Input data	Horizontal discretization
	Ground surface elevation
	Distribution codes for rainfall and meteorological stations
Interception	
Model parameters (Rutter Model) (for each crop type)	Canopy drainage parameters
	Canopy storage capacity (time varying)
	Ground cover indices (time varying)
Model parameters (K-J model) (for each crop type)	Leaf area index (time varying)
	Interception capacity coefficient
Input data	Rainfall rate
Evapotranspiration	
Model parameters (P-M equation)	Canopy resistance
	Aerodynamic resistance
	Ground cover indices (time varying)
	Ratio between actual and potential evapotranspiration as a function of soil moisture tension
	Root distribution with depth
Model parameters (K-J model) (for each crop type)	Empirical constants describing the ration between actual and potential evapotranspiration as function of soil moisture
	Leaf area index (time varying)
	Rooting depth (time varying)
	Root distribution coefficient
Input data	Meteorological data

Table 23.2 Continued.

TABLE 23.2. Continued.

Overland and channel flow	
Model parameters	Strickler roughness coefficients for overland and river flows
	Detention storage capacity on ground surface
	Coefficients of discharges for weir formulae
Input data	Specified levels and flows at boundaries
	Man-controlled discharges
	Topography of overland flow plane and river cross sections
	Riverbed lining thickness
	Riverbed lining permeability
Unsaturated zone	
Model parameters (for each soil type)	Soil moisture tension/content relationship
	Unsaturated hydraulic conductivity as a function of soil moisture content
	Maximum bypass ratio of net rainfall
Input data	Distribution codes for soil profiles
	Distribution codes for soil types in soil profiles
	Vertical node discretization in UZ
Saturated zone	
Model parameters	Storage coefficients
	Saturated hydraulic conductivities
	Drainage depth
	Time constant for drainage routing
Input data	Specified flows, gradients and heads at boundaries
	Location of abstraction or recharge wells
	Pumping and recharge rates
	Vertical node discretization in SZ
Snow melt	
Model parameters	Degree-day factor
	Snow zero plane displacement
	Snow roughness height
Input data	Meteorological and precipitation data

Data Overlays

A number of input data and parametric data values are required for each grid square in the horizontal grid. This is done as a stack of overlays containing codes describing structural data as well as data types. For each data type (e.g. soil type codes) a number of attributes (soil physical/properties) are attached specifying the actual properties of the data type.

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23.4. ADVECTION DISPERSION MODULE (MIKE SHE AD)

MIKE SHE AD simulates the detailed transport and spreading of dissolved conservative solutes from non-point and point sources. It may be coupled with chemical reaction modules to describe also the behaviour of non-conservative solutes.

The module describes the transport and spreading of solutes in the water flowing on the ground surface, in the soil water and in the groundwater by solving the advection- dispersion equation for the respective regimes.

In the following only the equations and the methodology is shown for the groundwater component. The methodology for the other components is very similar. Like for the WM module the largest problem encountered during the development of the AD module has been associated to the couplings between the various components.

EQUATIONS *(shown for groundwater component only)*

The transport of dissolved solutes in groundwater is described by solving the advection-dispersion equation numerically in a fully three-dimensional scheme.

The governing partial differential equation may be written as

$$\frac{\partial c}{\partial t} = -\frac{\partial}{\partial x_i}(cv_i) + \frac{\partial}{\partial x_i}\left(D_{ij}\frac{\partial c}{\partial x_j}\right) + R_c \quad i,j = 1,2,3 \qquad (23.11)$$

Symbol list

$c(x_i)$ - concentration of the solute [mg/l]

R_c - sources or sinks [mg/l/s]

D_{ij} - dispersion coefficients [m^2/s]

v_i - velocity components [m/s]

Important Parameters

The dispersion coefficients are the key parameters. They are determined from a number of dispersivities and the velocity components. The dispersivities have to be provided as input parameters.

Traditionally, the dispersion coefficients are assumed to solely depend on the longitudinal dispersivity, α_L, and the transverse dispersivity, α_T, but this assumption is only valid for isotropic conditions. Instead we suggest that the dispersion coefficients are calculated from four dispersivities so that a more correct description is achieved under anisotropic conditions.

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Solution Technique

The equation is solved numerically by an enhancement of the QUICKEST method originally introduced by Leonard (1979) for unsteady flows in one dimension. The scheme has been further developed to two and three dimensions by Justesen et al (1989), Ekebjerg and Justesen (1991) and Vested et al (1992).

The equation is formulated in an explicit scheme using upstream differencing techniques for the advection term and central differencing for the dispersion term. By developing the equations to third order the scheme becomes mass conservative and is computationally very efficient compared to implicit schemes. The numerical method is characterized by generating negligible numerical dispersion.

23.5. PRE- AND POSTPROCESSOR

The user interface of the MIKE SHE module includes powerful pre- and postprocessing facilities with particularly strong GIS capabilities. The software can be applied to the input data and results of all the MIKE SHE modules.

Some examples of the software capabilities are:

- Digitization of contours from maps such as those of the ground surface and geological layers
- Digitization of the river system layout
- Digitization of areally-distributed information, such as land-use, soil types, etc.
- Interpolation routines to provide point values and grid averages
- Graphical editing of 2-D data and river data
- Isoline plots
- Plots of the variations in space of a variable in any layer or along any line through the model
- Plots of time series of any variable

All graphical presentations can be in colour and are produced with a UNIRAS graphics package.

The user can easily design and produce animation of any variable. This provides a unique opportunity to present the dynamic behaviour of the simulated system and adds a new dimension to error handling and interpretation of results.

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23.6. APPLICATION EXPERIENCE

The MIKE SHE is today being used operationally by a large number of organisations in different countries ranging from university and research organisations to consulting engineering companies.

The original SHE version has been tested on a number of research catchments and applied to a few other projects, see e.g. Bathurst (1986), Refsgaard et al (1992) and Bathurst (this book) for further details.

MIKE SHE has as an extended version of SHE been applied to a large number of projects during the past few years. A list of applications in which DHI has been directly involved are shown in Tables (23.3) and (23.4) for research and consultancy projects, respectively. These applications illustrate the very wide range of water resources problems for which the MIKE SHE is a suitable tool.

TABLE 23.3.

List of MIKE SHE applications on externally funded research projects

Location	Project Title	Period	Topics
Denmark	Strategic environmental research programme	1993 -96	Groundwater pollution
EU	Development of a European Soil Erosion Model	1992 -94	Soil Erosion
Sweden	Effects of forestry drainage and clear-cutting on flood conditions	1991 -92	Impacts of human activity on floods
EU	Modelling of the nitrogen and pesticide transport and transformation on catchment scale	1991 -94	Agricultural pollution
Denmark	Research programme on groundwater pollution from waste disposal site	1988 -90	Groundwater pollution
Denmark	Research programme on nitrogen, phosphorous and organic matter	1986 -90	Simulation of nitrogen on catchment scale. Coupling with Daisy crop growth and nitrogen model
Denmark	Validation of pesticide models	1994 -96	Leaching of pesticides in clayey soils with preferential flow paths

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TABLE 23.4.

List of MIKE SHE applications on consultancy projects with DHI involvement

Location	Project Title	Period	Topics
Estonia	Tapa Airbase - Groundwater Model	1993	Groundwater pollution
Slovakia	Danubian Lowland - Groundwater Model	1992-95	Surface water quality, river and reservoir erosion and sedimentation, groundwater quality and geochemistry (redox), wetland ecology Coupling with ARC/INFO and Informix
Denmark	Four projects on Optimisation of remedial measures for safeguarding groundwater resources from pollution from waste disposal sites	1992-93	Groundwater pollution
UK	River Management Study, River Avon, Wessex	1992-93	Effects of groundwater abstraction and augmentation schemes on streamflow
Denmark	Environmental impact assessment of a highway construction	1992-93	Effects of groundwater drawdown due to tunnel construction
Australia	Irrigation salinity	1991-93	Process simulation of an irrigation district with focus on flow and salinity transport
Denmark	Identification of new well field for water supply	1991-92	Groundwater, effects of abstraction on streamflows and wetlands
Hungary	Assessment of pollution hazards in groundwater supplies	1991	Groundwater pollution
Denmark	Water supply planning in Aarhus county	1988-90	Groundwater resources, effects of abstractions on hydraulic heads and streamflows

In addition to the applications listed in the two tables MIKE SHE is used by other organisations to a large number of projects which are not known to the authors.

The SHE was originally developed with a view to describe the entire landphase of the hydrological cycle in a given catchment with a level of detail sufficiently fine to be able to claim a physically-based concept (Abbott et al. 1986a,b). The equations used in the model are with few exceptions non-empirical and well-known to represent the physical processes at the appropriate scales in the different parts of the hydrological cycle. The parameters in these equations can be obtained from measurements as long as they are compatible with the representative volumes for which the equations are derived.

In most regional catchment studies carried out so far, it has not been possible to represent the spatial variations in catchment characteristic with such a detail that the model could be considered physically-based. In fact, this was realised at an early stage when the applications changed from testing against analytical solutions and small scale research areas to applications on medium sized catchment areas.

However,it is the experience that the spatial resolutions and variations in properties used, provide a very good representation of the conditions in the areas modelled. In practise the spatial variations are derived from maps describing topography, soil and land-use pattern and interpreted geological conditions, combined with information about the general properties of the different map units. The model parameters values are then modified during calibration to match observed conditions at discrete points.

There are a number of fundamental scale problems which needs to be carefully considered in the model applications. This is particular important when describing the interaction between thesurface flow and the subsurface flows. A few areas where scale problems are encountered include:

- The interaction between groundwater and river. Since the flow is based on Darcy's law using the gradient between the river water level and the groundwater heads in the adjacent grid squares, the flow rates and the resultant head changes will depend on the spatial resolution used. This is an important aspect in for example simulating the hydrograph recessions correct.

- In catchments with a dense drainage network it is often not possible to represent entire drainage system (many streams are of ephemeral nature). For such situations subgrid variations in the topography need to be accounted for in order to simulate the hydrograph response in the main streams correctly.

- For modelling of infiltration and vertical unsaturated flow in the soil, the hydraulic parameters used in Richards' equation can be obtained from laboratory measurements on small undisturbed soil

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samples. However, for grid squares covering large areas (eg. 25 ha) these are seldom representative unless completely homogeneous conditions exist in the horizontal directions. Therefore effective or representative parameters are used, which means that the simulated soil moisture conditions can not be verified directly.

In fact much of the criticism against MIKE SHE arise often from the way the unsaturated flow is simulated and very seldom on how the groundwater conditions are treated.

For most catchment simulations the use of Richards' equation becomes conceptual rather than physically based and simpler approaches could be chosen. Nevertheless, this equation provides a good routing description, and the capability to simulate capillary rise under shallow water table conditions is an important option for studies where eg. wetland areas are included. For situations where Darcy law' does not apply a simple macropore option is included in the solution.

Because representative parameter values are used, the reliability of the results depends very much on the data available for comparison of the simulated spatial and temporal variations with observations. This is well-known from groundwater applications, where the aquifer properties (conductivities or transmissivities) are derived based on calibration against observed head variations in discrete points. For regional catchment studies themodel performance is usually evaluated based on comparisons against river discharges and groundwater heads. Very seldom measured soil moisture data are available, and if they are such comparisons will require that the site specific properties are known.

It is often stated that distributed models requires a large number of data and therefore very time consuming and complicated to set up and calibrate. In fact, a number of short-term screening evaluation projects have been carried out with MIKE SHE for example in connection with studying the contamination risks from waste disposals. In these studies only sparse existing information about the hydrogeological conditions was available. The model was used to obtain an improved knowledge about the possible flow patterns around the waste disposal site based on the existing geological interpretations. These applications could also be used to identify where existing knowledge is lacking and assist in defining appropriate monitoring programme.

Another common argument against distributed models is the risk of over-parameterization. This risk is of course always there. However, the general experience is that if the data are lacking to describe the spatial variations in the catchment it is too time-consuming and not worthwhile to modify a large number of parameter values in order to improve eg. hydrograph predictions. In such cases very few parameters are used in the calibration and the reliability of the results are evaluated with this in mind.

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23.7. ONGOING RESEARCH

From the above application records it appears that MIKE SHE already has been used comprehensively both for research studies and for practical routine applications. These applications reflect that for certain types of studies there is no adequate alternative to an integrated, distributed, physically-based modelling approach like the MIKE SHE. Nevertheless, it is realized that MIKE SHE, in its present form, is far from being complete as a tool for advanced hydrological analyses. Many both practical and fundamental problems need to be solved through future research activities. A very significant part of the research carried out in these years in the international scientific community is in fact of direct relevance and most valuable in this context.

At DHI research and developments related to MIKE SHE is carried out in the following fields:

- Improvement of process descriptions. Research work on macropore flow and solute transport is presently undertaken. Other activities such as inclusion of density effects in the groundwater component and description of hysteresis phenomena in the unsaturated zone are planned in the coming years.

- Improvements of numerical efficiency is continuously taking place.

- Interface to geographic information systems (ARC/INFO) is being developed.

- Coupling with DHI's generalized river modelling system (MIKE 11, see Chapter 21 of this book) is going on. A coupled MIKE SHE/MIKE 11 enables description of sediment transport and water quality processes also in the river system as well as description of complex river and canal systems. Typical area of application is irrigation command areas, where networks of both irrigation and drainage canals exist together with a large number of different hydraulic regulating structures.

- Fundamental research on establishment of rigorous methodology on parameterization, calibration and validation is urgently needed. Some first, small steps have been taken as described in Refsgaard et al (1992), Jain et al (1992) and DHI (1993).

- Fundamental research on scale problems related to spatial variability of hydrological parameters is urgently needed. In particular problems related to different scales used for data sampling, process description and model discretization need to be addressed. Although comprehensive international research is carried out in these years no operational results and conclusions are evident.

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23.8. CASE STUDY: MODELLING OF NITROGEN TRANSPORT AND TRANSFORMATION ON A CATCHMENT SCALE

23.8.1. Introduction

The present case study is one of the outputs from a comprehensive Danish research and development programme (1986-90), which was carried out with the aim of studying the pollution from nutrients and organic matters in agriculture. The research programme was multidisciplinary and involved a large number of research institutions. It included field investigations, process studies and modelling.

The present case study briefly describes a distributed hydrological modelling of nitrate transport and transformation for the 440 km^2 Karup River catchment. The nitrogen modelling covers the entire land phase of the hydrological cycle - from the source on the soil surface, through the soil zone and the ground water to the streams. The modelling was based on the MIKE SHE CN, which is a combination of the MIKE SHE WM + AD modules and the DAISY model for simulation of the nitrogen dynamics in the root zone.

A more thorough description of the DAISY model is given by Hansen et al (1991), whereas a detailed description of the present case study is presented by Styczen and Storm (1993).

23.8.2. Model setup

The Karup catchment was represented in a three-dimensional network. The discretization is 500 m in the horizontal directions and varies in the vertical from 5 to 40 cm in the unsaturated zone, and 5 m in the permanently saturated zone. Information on soil and vegetation properties were collected, and from the information from a number of wells, a three-dimensional geological map was superimposed on the model grid to provide the hydrogeological parameter values. The topography and the river network have been digitized, and all relevant climatological data collected. The overall land use has been identified. Some of the data and parts of the MIKE SHE model setup are shown in Figs. 23.5 - 23.9.

23.8.3. Results

Discharge and groundwater table hydrographs

The streamflow is simulated for the period 1969 - 1988 at several sites. A comparison with measured discharge at the catchment outlet is shown for

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four years in Fig. 23.10. In addition the simulated groundwater table is compared with observations in selected wells (Fig. 23.11). The comparison shows that the modelling system simulates the hydrological regime with acceptable accuracy.

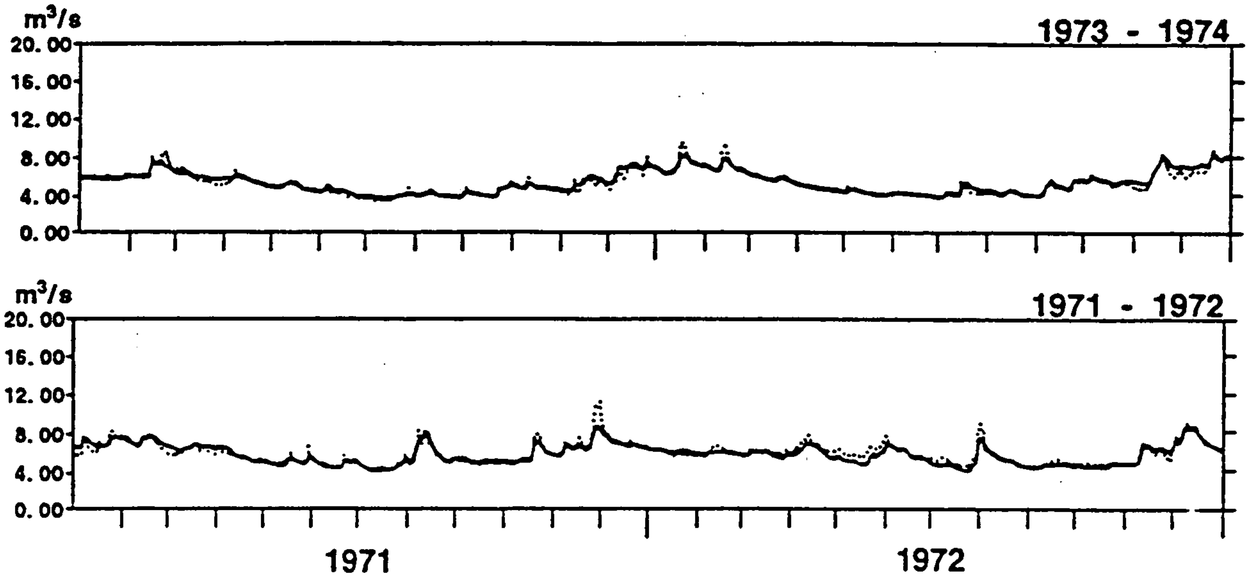

Fig. 23.10. Comparison between simulated and observed river runoff for the period 1971-74.

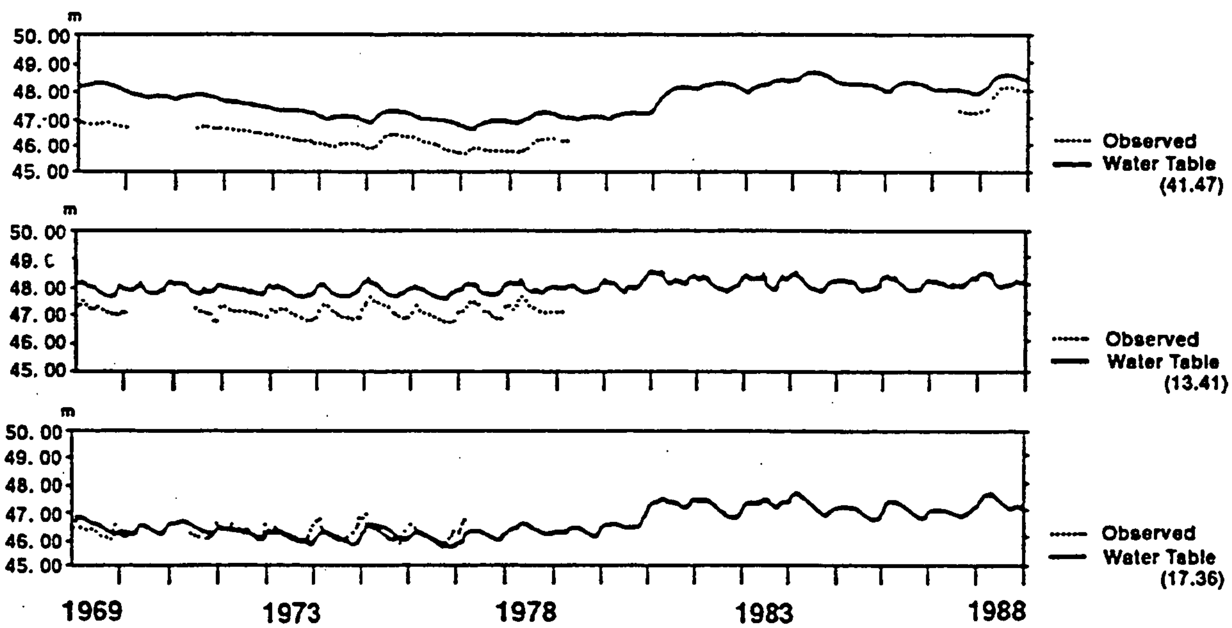

Fig. 23.11. Comparison between simulated and observed groundwater table time series in selected wells.

Leakage from the Root Zone

To simulate the trend in the nitrate concentrations in the ground water and the streams, it is necessary to have information on the history of the fertilizer application in space and time. This information is difficult to obtain in details, for example it is not possible to estimate which type of crop was growing on one particular field in one particular year in the past. The most detailed information one can expect to obtain is an areal percentage of the various crops, and the types of farming practices that have been carried out in the area. Based on this information a series of 14 crop rotation schemes covering the period of interest was established, and at random distributed over the area.

Based on estimated application rates of organic and mineral fertilizer to the individual crops each year, the DAISY model simulates the crop growth, root uptake, mineralization and leakage of nitrate from the root zone. Figure 23.12 shows time series of application and leakage for selected crop rotation schemes. On farms which are based on mainly meat production a large amount of organic fertilizer will often be applied on the fields in the autumn. In this period there is a potential risk for significant losses to the groundwater system.

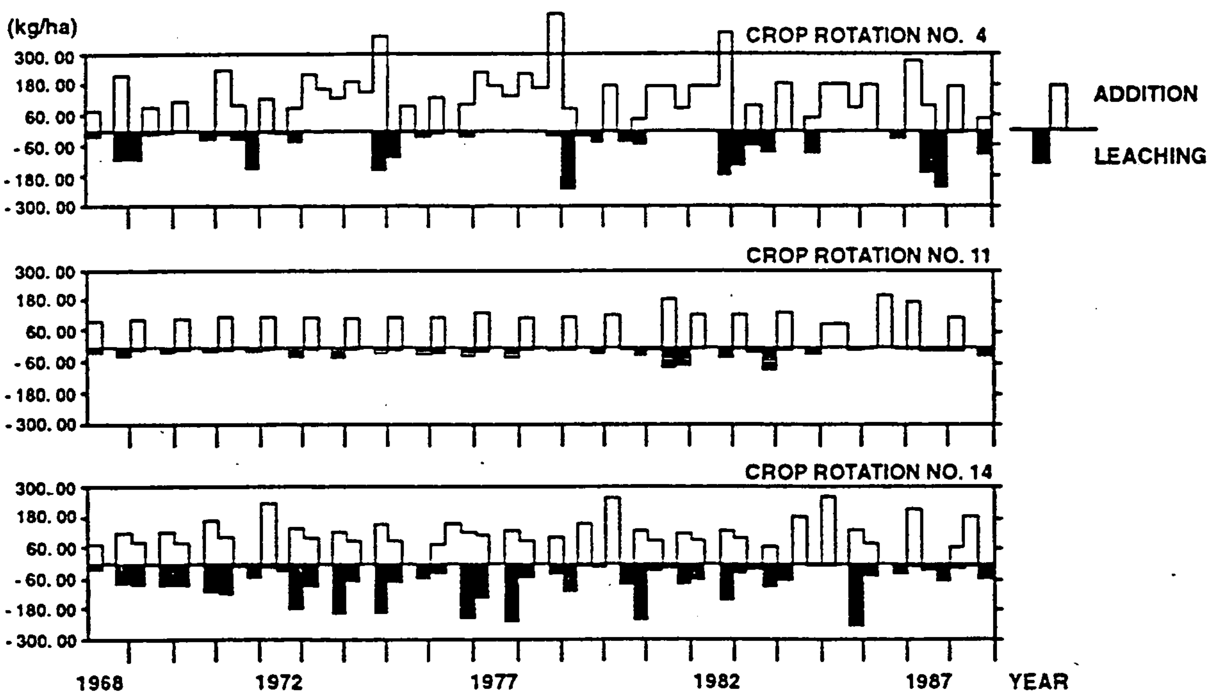

Fig. 23.12. Nitrate leaching ($NO_3^- - N$) from three of the crop rotations calculated by DAISY and summarized over four-months periods. The shown additions of N only include mineral fertilizer and the already mineralized part of manure.

Nitrate concentrations in groundwater

While the root zone model simulates one 'soil column' at a time the total model allows studies of the variations in space and time at regional scale. Fig. 23.13 illustrates the variation in simulated NO_3^--concentrations in the upper groundwater layer of the Karup catchment below three selected cropping schemes for two points with different depths of the unsaturated zone. A deep unsaturated zone is seen to dampen the influence of a single year.

Figure 23.14 shows the spatial variation in simulated NO_3^--concentrations in the upper groundwater layer at a specific time. The very large variation of concentration both in space and time is noticed.

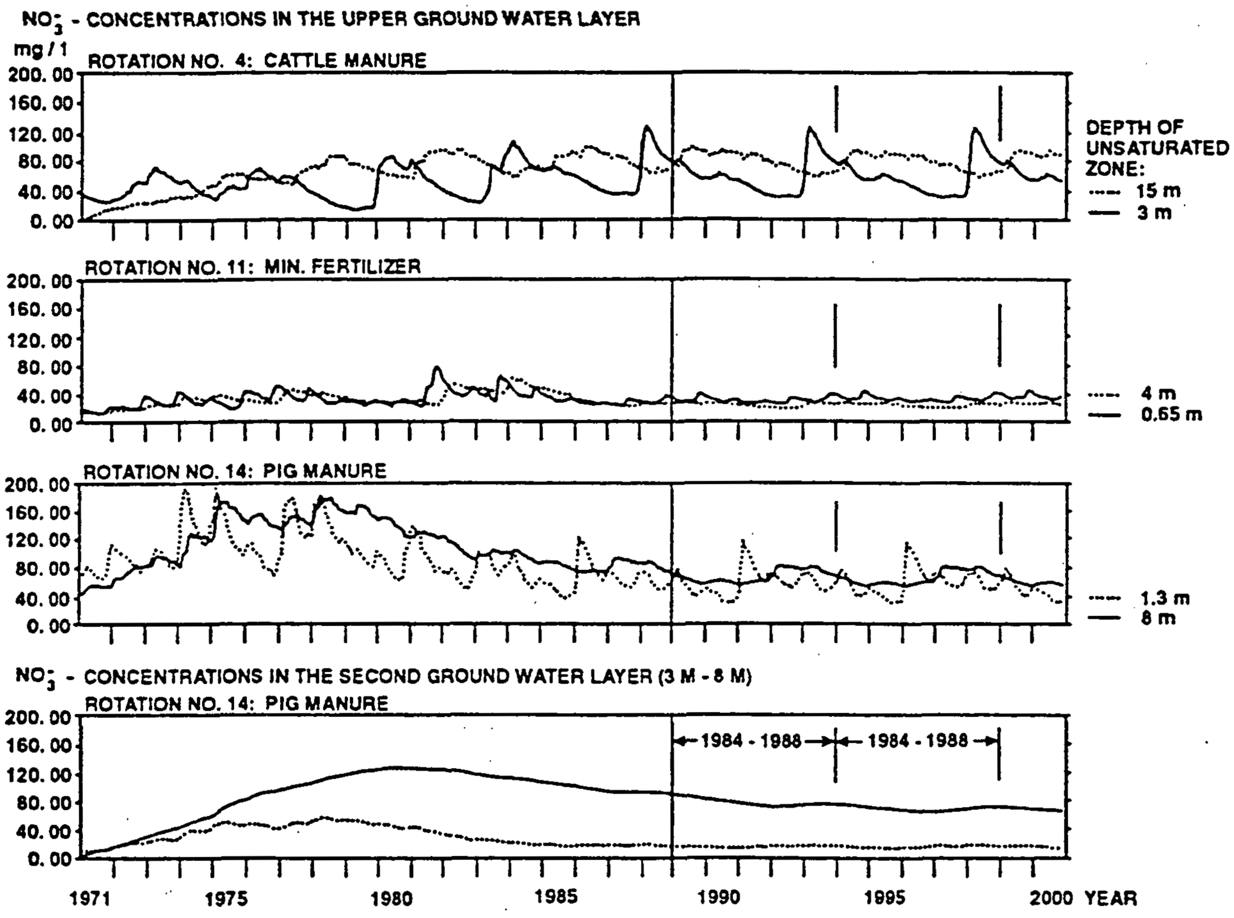

Fig. 23.13. Temporal variation in NO_3^--concentrations in the upper groundwater layer beneath three selected rotation schemes, with two different distances to the groundwater table. The data are extracted from selected grids (not averaged).

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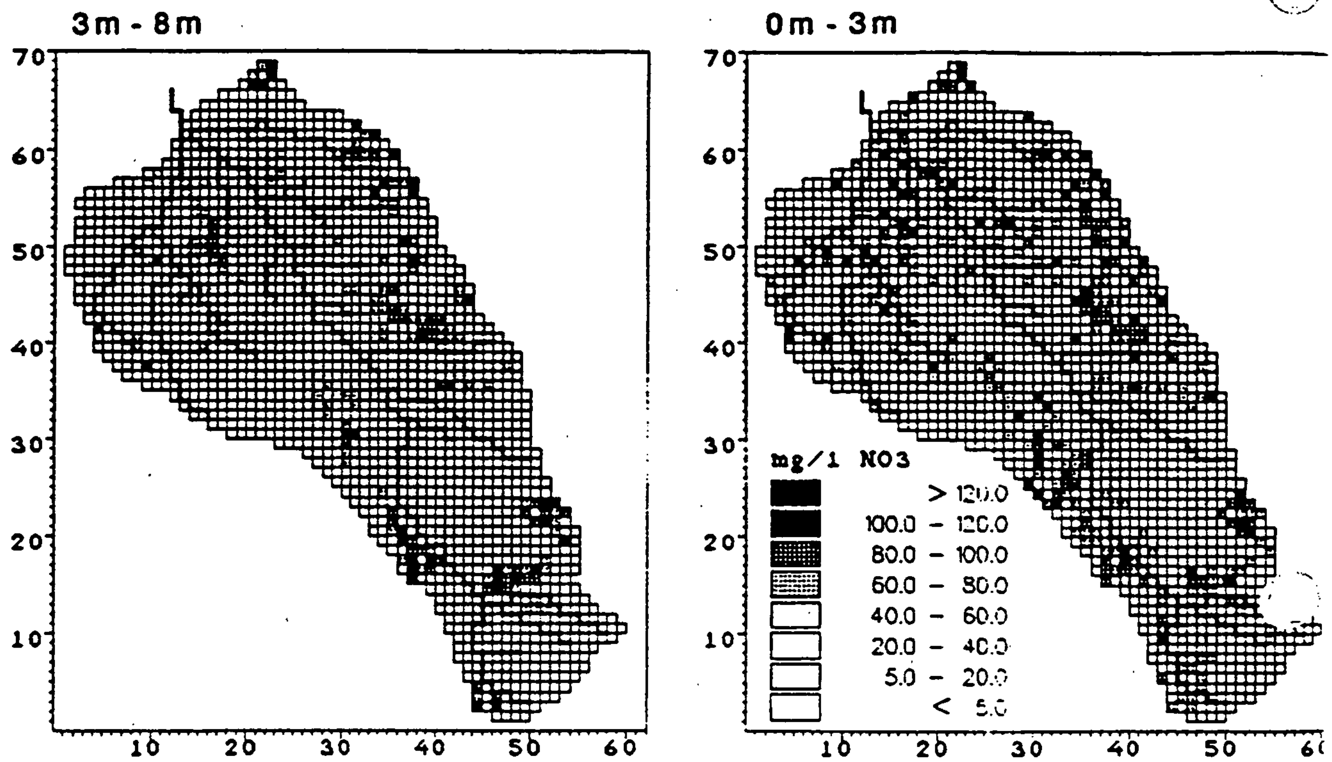

Fig. 23.14. Spatial variation in NO_3^--concentrations in the upper groundwater layer over the entire catchment at a specific time.

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DHI ref. 68/95

MODELLING ENVIRONMENTAL CHANGE IN THE WAKOOL IRRIGATION DISTRICT

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Abstract This paper describes the development and application of an integrated catchment model for the Wakool Irrigation District in New South Wales Australia. The WID, which is bounded by the Wakool River in the south and Edward River in the north, has experienced high watertables and increasing land salinisation since the mid 1950s. In 1980, an area of 30,900 ha within the district had shallow watertables (less than 2 metres below ground surface). This area has since been reduced through operation of 59 pumpsites involving transfer of approximately 12,500 ML/year to two evaporation basins. Nevertheless, the watertable is still rising at an average rate of 30-70 mm/year due to large-scale irrigation. Community concerns about the expanding high watertable areas, land salinisation, the role of groundwater pumping, and the general sustainability of the area has led to the development of a Land and Water Management Plan (LWMP) for WID. The main objectives of the LWMP includes issues such as improvement of productivity, education and encouraging landholders to adopt sustainable land and water management practices, decrease and control of the rise in watertables, and monitoring the progress of subsequent strategies. An important component in the development and execution of the LWMP is to establish a hydrological model to analyze the complex hydrological regime in the WID, and predict the environmental impacts of various planned management options. This paper presents the Wakool Irrigation District Model (WIDM) based on the MIKE SHE integrated catchment modelling system. The WIDM provides a complete description of the complex hydrological regime in WID involving temporal and spatial variations in the exchange of water between the ground surface, drainage and supply systems, and the groundwater aquifers within the area. Management options have been analyzed for a timeframe of 30 years and include scenarios which focus on the surface water regime (extension of the drainage system and/or sealing of supply channels) as well as the subsurface water regime (shallow and deep groundwater pumping schemes).

1. INTRODUCTION

Rising groundwater and subsequent land salinisation have been observed during the last three decades in many Irrigation Districts throughout south-western part of NSW. Crop yield is significantly influenced by shallow groundwater tables lying within a critical depth (less than 2m) below the land surface. Concerns about the increasing environmental problems, have encouraged communities to develop a Land and Water Management Plan (LWMP) to ensure a future sustainable agricultural industry. The various management options proposed in the LWMPs depend on the individual characteristics of the districts, and includes common features concerning surface drainage, infrastructure, flood plain management, sub-surface drainage schemes, on-farm management etc.

Many hydrological and hydrogeological studies have been carried out to increase our understanding of the hydrological systems and assess future conditions in the districts (e.g. Bogoda et. al. [1994], GHD [1995]. These include development of a comprehensive hydrological model, which may be a valuable tool not only in the planning phase, but also during implementation and operation of selected options.

This paper presents the development and application of an integrated surface and subsurface model for the WID. The overall objective of the study was to develop a reliable tool, which can predict the environmental impact of implementing various management options for the WID.

2. DESCRIPTION OF THE AREA

The Wakool Irrigation District, which covers an area of 250,000 ha, is located in the central part of the Murray Darling River Basin west of the town Deniliquin and north-east of Swan Hill. The district is bounded by the Wakool River in the south and Edward River in the north (Figure 1). Irrigation commenced in WID in 1936 after the completion of Stevens Weir on the Edward River. In the beginning irrigation was applied only in limited areas on natural pastures, but the irrigated area has since expanded substantially and covers today approximately 220,000 ha. The major agricultural products includes rice, cereals, wool, milk and meat.

The climate in the district is semi-arid with an average rainfall of approximately 360 mm per year. High rainfalls occur mainly during the winter months June to August, and a large part of the district is occasionally prone to flooding. The topography is in general flat with a westerly direction of groundwater flow and natural surface drainage.

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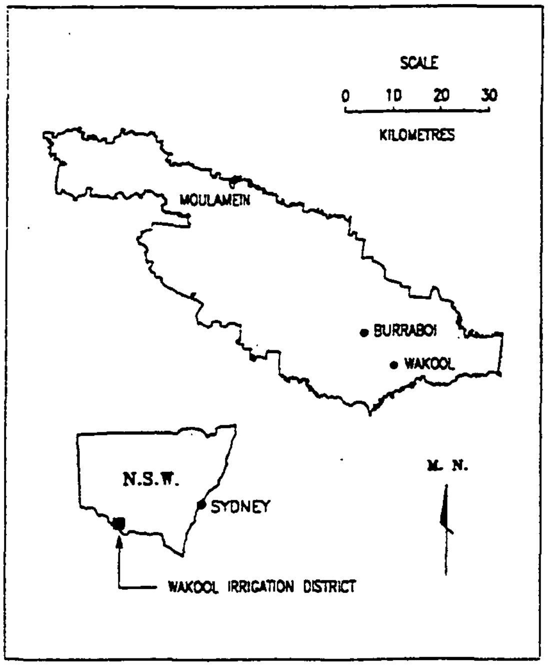

Figure 1: Location of the Wakool Irrigation District

Three major hydrogeological formations are located within the area with the Shepparton Formation forming the upper aquifer system. The depth and extent of the water bearing layers in the Shepparton are limited with pronounced variations in transmissivity across the district. So called prior stream and ancestral river systems, which are remnants from former river routes, play an important role for the groundwater flow paths. The deep regional aquifer systems, the Calivil and Renmark Formations play an important role for the regional groundwater flow in the Murray Basin, and the piezometric heads in these aquifers controls the exchange to and from the Shepparton.

In the WID, there seems to be a fine balance between heads in the Shepparton and the Calivil/Renmark aquifers, which determines the leakage exchange between the aquifers. In contrast to more easterly located irrigation areas, the piezometric heads in the deeper aquifers has reached the same overall level as the groundwater table, and therefore decreased the potential for deep recharge within the area.

In general, the Shepparton provides a relatively poor drainage capacity for the infiltrating water through the rootzone. The natural rainfall combined with large-scale irrigation in the district have therefore caused the area of shallow water table to increase from 7,200 ha to 47,500 ha during the period 1960-75. A comprehensive groundwater pumping scheme, the Wakool Tullakool Sub-Surface Drainage System (WTSSDS), established in the early 1980's has successfully alleviated the problems in the south-eastern part of the district. However, currently an area of 26,100 ha is affected by high watertable.

The current irrigation practices and possible further rise in piezometric head in the deeper aquifers are expected to contribute to a further increase in groundwater tables in many parts of the district.

3. METHODOLOGY

The development of the Wakool Irrigation District Model has involved six steps:

- Identification off all relevant information;
- Collation of information and data processing;
- Development of a conceptual hydrological model for the area
- Development of the numerical model;
- Calibration of the model;
- Prediction of the environmental impacts of proposed management options

Each of the above steps is briefly described below

3.1 Model Input

A common belief is that only very sparse information is available to support a comprehensive model development for the irrigation districts in the Murray Region. This may be true considering direct usable information only. However, at least for the WID, considerable indirect information has been available, which after processing has been beneficial for model development. A broad list of the most important information utilized in the development of the model is presented in Table 1.

3.2 Model Conceptualisation

In order to develop a reliable modelling tool, the mathematical model should reflect all the importan natural processes as well as provide the possibility to impose relevant human interventions in the natural system. Important features, which are common for irrigation areas include: natural and constructed drainage lines; supply channels with particular emphasis on a representation of sites where significant seepage losses may occur; pumping schemes, and reallocation of saline groundwater to evaporation basins.

The model must further be able to simulate the infiltration and evapotranspiration for agricultural crops on specific soils. In this respect the capillary rise mechanism in shallow watertable areas is an important factor in the overall and local water balances.

Table 1: Major Information used in the model application

Information	Distribution	Comments
Rainfall and water usage	T & S[1)]	Water usage for each crop type has been estimated (1975-91)
Potential Evapotranspiration	T	-
Soil Distribution	S	Properties have been estimated from grain size distributions
Land Use	S	Obtained from remote sensing for year 1990
Hydrogeological Characteristics	S	Well log information
Topography	S	
Supply channel network	S	Total seepage loss of 20 000 Ml/year is imposed at selected locations in the model
Drainage network	S	Both natural and man-made channels included
Groundwater Pumping	T & S	The scheme started in 1981, the current operation is 13 000 Ml/year
Groundwater table observations	T & S	Half-yearly data for approx. 1200 bores

[1)] T: temporal variations, S: spatial variations

The groundwater system in the WID area can be divided into two aquifer systems, Shepparton and Calivil/Renmark. The former is further divided into two layers, representing a sandy and relatively permeable layer in the upper 20 meters. We assume that the main groundwater movement in horizontal directions within the area occurs in this layer. Beneath it is the low permeable Lower Shepparton, which primarily is responsible for the vertical flow exchange between the Upper Shepparton and the Calivil/Renmark system. Within the Upper Shepparton layer, the prior and ancestral streams may form significant routes for groundwater flow.

The Calivil/Renmark Formations extend beyond the model area. The available hydrogeological information is in general very sparse for these formations, and refinements in the calibration of the deeper aquifers will be required when additional information becomes available. However, since the flow exchange between these and the Shepparton is an important factor, they have been included in the model.

3.2 The Numerical Model

An integrated catchment modelling system (MIKE SHE, DHI [1993], Lohani et. al. [1993], Carr et. al. [1993]) has been adopted for this study. MIKE SHE was chosen because it represents all the major flow features in the area and is well-suited to describe the dynamic interaction between the surface and subsurface water systems, which in particular is important in the high watertable areas of the district.

The use of MIKE SHE has allowed us to analyze the wide variety of localized management options, which involves not only the groundwater system, but also changes on the landsurface such as: sealing of irrigation distribution system, extension of drainage network, on-farm practices etc. Flooding events from the Wakool and Edward rivers have not specifically been introduced in the model, but the effect of local flooding has been analyzed.

A finite difference grid with a spatial resolution of 4 km^2 represents the spatial variations of catchment characteristics, inputs and imposed stresses. The drainage system is described with the river module, whereas the supply system is represented only at reaches at which seepage losses occur.

In the Upper Shepparton, groundwater flow occurs across the eastern and western boundaries, but not along the Edward and Wakool rivers. Fixed head conditions have been applied along the boundaries in the Calivil/Renmark layer assuming that future changes in the Calivil/Renmark arises mainly from the head difference between the Shepparton and the Calivil/Renmark aquifers.

The model has been run with variable timesteps depending on the actual conditions. Because the model has to calculate overland flow and infiltration rates at the ground surface, it has been necessary to apply very small timesteps.

3.3 Model Calibration

The model was calibrated on the period 1980 to 1990. This period includes a mixture of dry and wet years without any major impacts of flooding and therefore the most suitable period for the model calibration. Historical measurements of groundwater levels in more than 1200 bores screened in the sandy layers of the Shepparton Formation were used. Insufficient data on streamflows and soil moisture was available for calibration of the channel and soil moisture components of the model.

The main emphasis of the calibration has been to achieve a level of model accuracy that will provide good temporal and spatial agreements between simulated and observed groundwater levels throughout the irrigation district. The performance in calibration can be measured by the model's ability to predict historical regional groundwater flow patterns and historic trends in groundwater levels.

To certify that the model also simulates the surface flow within the district correctly will require additional calibration of streamflow at a number of locations e.g. Wakool River, Edward River, Niemur River, Jimaringle Creek. The lack calibration data for the streamflow simulation does not seriously influence the estimation of groundwater recharge and groundwater levels. However, it can to some extend influence the reliability of simulated management options related to the surface water system. A general overestimation of certain input data, e.g. irrigation applications may result in an overestimation of streamflows and/or overestimation of the simulated actual evaporation, but have only little influence on the simulated groundwater recharge.

Figure 2 shows a comparison between measured and simulated groundwater levels in selected bores. Inspection of maps with simulated and observed contours revealed that both spatially and temporally the model is able to achieve good agreement with the observed groundwater levels. The very detailed ground water monitoring in the district provides a good description of the spatial variations. However, as seen in figure 2, that detailed monitoring in selected key bores would be beneficial to check the watertable fluctuations in detail.

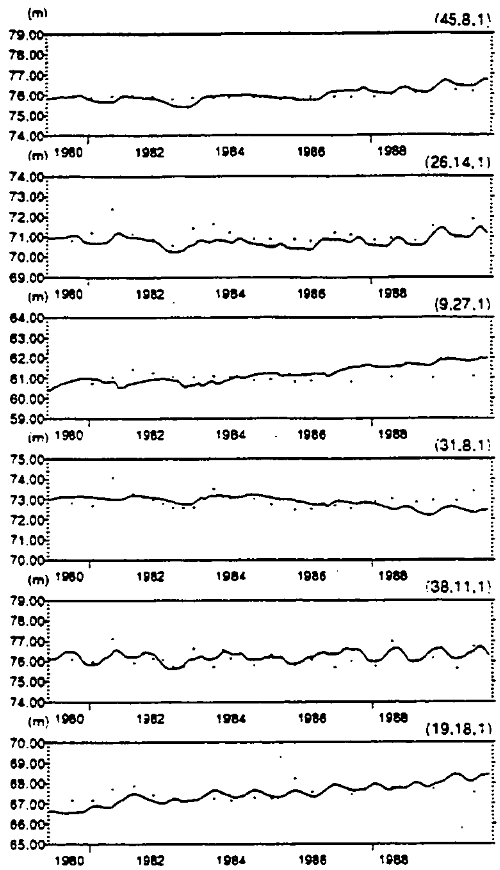

Figure 2: Comparison of measured (dots) and simulated (full line) groundwater tables in selected bores.

From Figure 3 it appears that the model is able to describe the overall groundwater levels in the WTSSDS very well. Some of the deviations, in particular around the pumping wells, arises from the relative coarse spatial resolution used in the model. If a sub-model for the WTSSDS was established these variations would be improved significantly.

It is worth noting, that simulated groundwater mounds along sensitive reaches of the supply channels are confirmed by the observations. Also the 'tongue' of high groundwater which extend along the southern prior stream channel is simulated very well.

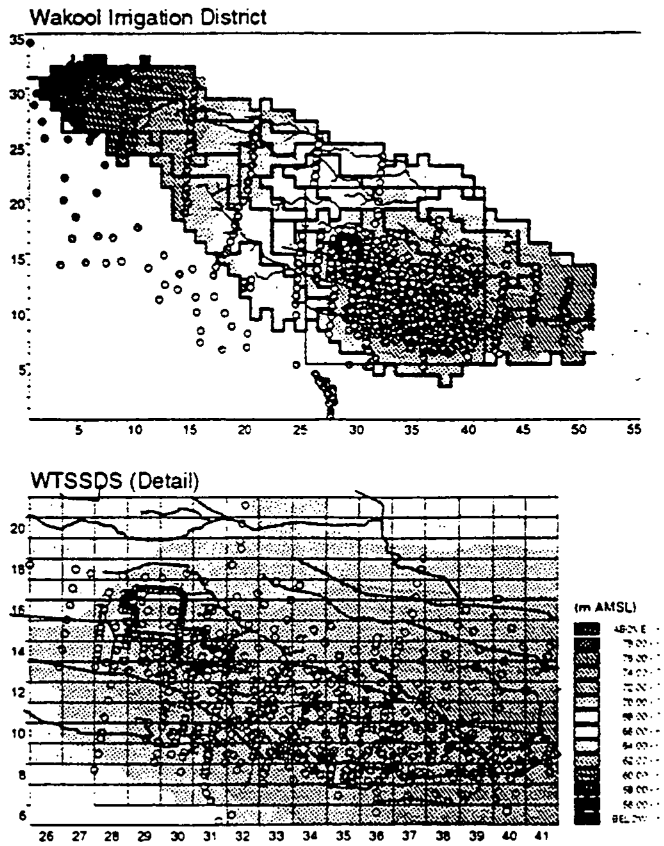

Figure 3: Comparison of observed (circles) and simulated groundwater heads in Wakool and in the WTSSDS area in 1988.

3.4 Predictions of environmental Impacts

A number of possible strategies, which have been proposed in the Land and Water Management Plan for Wakool have been modelled. The model scenarios include:

- No Plan Scenario;
- Additional shallow pumping wells in the Shepparton Formation;
- Deep pumping wells in the Calivil Formation (in total 21,000 Ml/year);
- Sealing of the supply system at selected sites.

The predictions are carried out for a thirty year framework to year 2020. A key measure to check the success of the proposed strategies is to analyze the total extent and distribution of the shallow watertable areas (depth < 2meter). Figure 4 shows the evolution of shallow watertable

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further actions are taken and for the case where the shallow pumping is increased respectively.

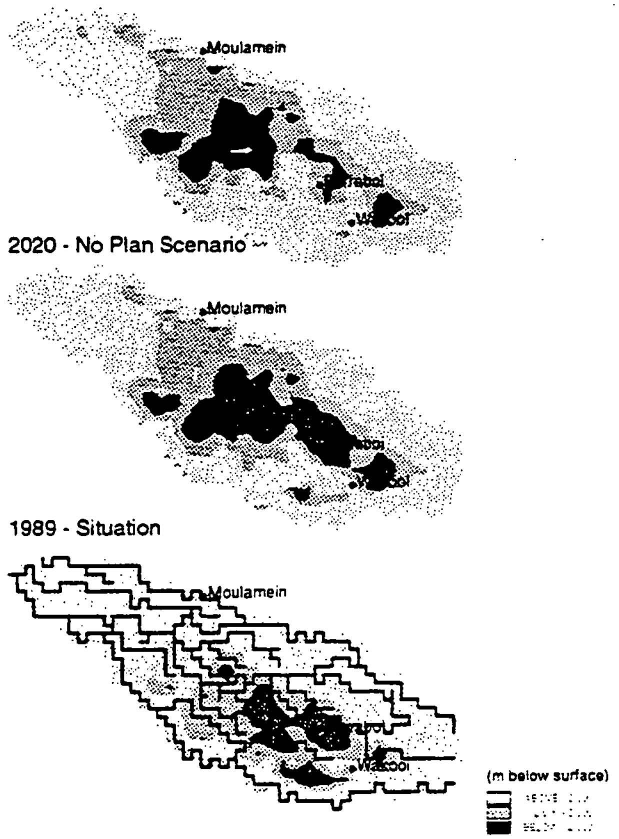

Figure 4: Areas of shallow watertable in 1989 and predicted for year 2020 for two scenarios.

It appears from Figure 4 that if no remedial actions are taken, we can expect the area with shallow watertable to increase further. The current subsurface drainage scheme (WTSSDS) controls effectively the watertable in the area covered by the pumping scheme. However, there is a clear trend of rising water table in the north-western direction of the district. The individual pumps located in the Shepparton Formation seems only to have local effects. To control a larger area in its existing form will therefore require a scheme consisting of a large number of pumps.

The introduction of additional 48 pumps located mainly inside the WTSSDS area have some effects locally, but can not successfully prevent a further rise in central parts of the district.

Punthakey et. al. [1994] showed that optimization of the existing pumping scheme and determine optimal locations for new pumpsites could conceively protect a much larger area then presently covered by the existing scheme.

The deep pumping scenario may alleviate the problem in large areas. Depending on the magnitude and location of the pumps, deep pumping can diminish the current watertable rise because a downward leakage from the Shepparton Formation to the Calivil Formation will be generated. It seems that deep pumping from the Calivil may influence larger areas in the Shepparton, but to a lesser extent than direct pumping from the Shepparton. Pumping in Calivil can also have some regional impact on areas outside the WID.

Figure 5 shows the effect of deep pumping on the local water balance for the WTSSDS. It is seen that the deep pumping generates a downward flow from the Shepparton to the Calivil aquifer, but also a considerable net inflow from outside in the deep aquifer.

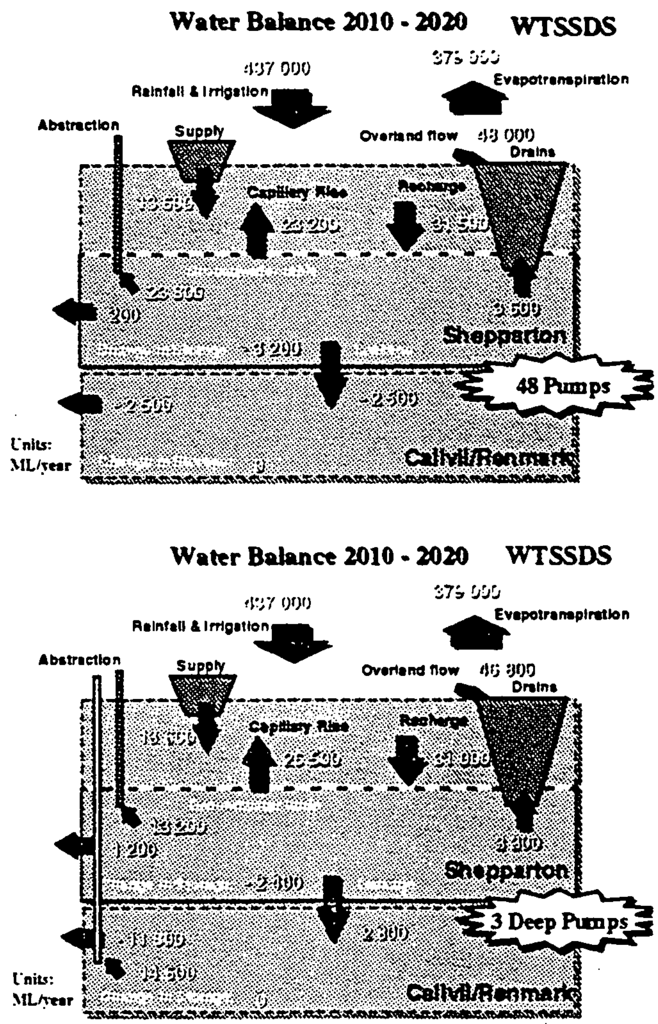

Figure 5: Water balances for the WTSSDS area for the "Shallow Pumping" and the "Deep Pumping" scenarios.

4. CONCLUSIONS AND RECOMMENDATIONS

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Integrated catchment modelling of the hydrological regime in irrigation districts, suffering from high watertable and land salinisation, is an important approach for managing current problems within irrigation districts, and to evaluate the effectiveness of proposed options. In this modelling study a range of strategies proposed in the Land and Water Management Plan for the Wakool Irrigation District has been tested.

The model predicts that with the current irrigation practices the area with shallow watertable will further expand, and by year 2020 cover large areas of the central part of WID. Both shallow and deep pumping schemes will alleviate the problems, but these have to be designed carefully in order to obtain full advantage.

Once proposed plans are being implemented, the calibrated model may be useful in connection with detailed designs. In conjunction with appropriate monitoring programs the model may in the future provide a good tool to ensure envorinmental problems within the district as well as for downstream users can be solved in time.

5. ACKNOWLEDGMENTS

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MODELLING LEACHING OF PESTICIDES

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1. INTRODUCTION

In Denmark 99 % of the drinking water is derived from groundwater. This implies that efforts towards improved groundwater protection has high priority among the Danish authorities. Recent findings of pesticides in drinking water supply wells, in streams and in shallow groundwater have set focus on the methods and assumptions used for pesticide registration by the Danish Environmental Protection Agency (NAEP). In particular, the appearance of phenoxyacids in many well borings from a few meters up to 80 meters below the soil surface, and in the aquifers below thick moraine clay layers has cast doubt on the applicability of the registration procedure. Phenoxyacids were considered relatively safe due to a short estimated degradation time /10,25/.

The NAEP has based the registration criterias on the EEC directive 80/778 on Drinking Water, and on Statutory Order No. 6 of January 4, 1980, on Water Quality and Supervision of Water Supply Plants. The concentration limit for pesticides is 0.1 μg/l for each of the pesticides, and 0.5 μg/l for total amount of pesticides. The registration procedure regarding groundwater contamination risk is based on determination of pesticide mobility in soils described by adsorption and degradation characteristics which are derived from laboratory and lysimeter/field tests on at least three soil types. If the active ingredient following a laboratory test leaches in quantities exceeding a certain percentage of the applied amount, supplementary tests must be conducted to assess the mobility under more realistic conditions such as lysimeter or field tests.

In the light of the many unexpected findings of pesticides in the groundwater, one could argue whether the chosen tests are realistic with respect to the hydrological conditions prevailing in Denmark. Certainly, fluctuating groundwater tables, artificially drained areas and the presence of macropores in many Danish soils must influence the probability of pesticides reaching the groundwater.

Pesticide properties and hydrological conditions may vary in space. A fast degradation rate determined in the upper part of the unsaturated zone may not be valid if the substance is washed down through macropores shortly after application. The degradation in deeper layers of the unsaturated zone has been found to occur at a much slower rate /2/. Additionally, very little is known about the rate of degradation occurring in the deeper saturated zone, though it is expected to be negligible for most pesticides due to the anoxic conditions present in this area.

Another aspect influencing the risk of pesticide leaching is the time varying interactions between the pesticide properties and the hydrological conditions. Pesticides possessing a combination of properties which is expected to minimize the risk of leaching under conditions characteristic for spring application, may perform totally different if applied during the autumn.

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A way of taking these aspects into account in the registration procedure would be to use dynamic numerical models describing pesticide leaching. The major benefit of models are their ability to take into account the variation in time and space, under a more diverse range of conditions than can be produced in laboratory or field experiments, due to either technical or economical limitations. However, in order to be able to rely on the simulation results, it is crucial to investigate and take into account the limitations of a certain model with respect to applicability of the process descriptions included.

At the moment the Danish authorities are not using models in the registration procedure, but as a result of the EEC directive 91/414, and the directive presently being prepared regarding uniform principles in the registration, mathematical models must be incorporated into the registration procedure in the future.

The objective of the present project was to evaluate a number of existing pesticide models with respect to their ability to predict water and pesticide movement under the hydrological conditions characteristic for Danish soils.

2. MATERIALS AND METHODS

2.1 Model description

From existing and available pesticide leaching models eight models were considered for this evaluation: Chemical Movements in Layered Soils (CMLS, version 4.0) /26,27/, Ground-water Loading Effects on Agricultural Management Systems (GLEAMS, version 1.8.54) /24/, Leaching Estimation and Chemistry Model (LEACHM version 2.0) /33,34/, Method of Underground Soil Evaluation (MOUSE, version 7x) /29,31/, Pesticide Root Zone Model (PRZM, Release 1) /11,12/, Root Zone Water Quality Model (RZWQM) /15/, Pesticide Leaching and Accumulation (PESTLA, version 1.1) /5,7,8/, A Model of Water Movement and Solute Transport in Macroporous Soils (MACRO version available in 1992) /19/.

Of these models PESTLA is presently being used for registration in Holland, and a modified version of PRZM (PELMO) is used in Germany.

All the models are dynamic, deterministic and one-dimensional in their approach of describing water and pesticide behaviour in the root zone and the unsaturated zone. However, they differ with regard to complexity. An overview of the process descriptions used in the eight models is presented in Table 1.

Table 1: Model contents and approaches (2 pages)

Contents	CMLS (USA)	GLEAMS (USA)	LEACHM (USA)	MOUSE (USA)	PRZM (USA)	RZWQM (USA)	PESTLA (HOLLAND)	MACRO (SWEDEN)
Water flow	Piston displacement of water, instantaneous redistr. betw. FC and WP.	Predicts water flow betw. soil layers based on storage *similar to the "tipping bucket"* method	Solves Richards eqn.	Water drainage occurs under unit hydraulic gradient. K(*0*) is calc. by exp. function	"Tipping bucket", Operates between FC and WP. Inst. or time dep. water redistr	Layered Green and Ampt inf. from the soil surface and radial inf. from macropores. Richards eqn. for redistr. betw. layers	Solves Richards eqn.	Solves Richards eqn. in matrix. Solves Darcy's eqn. in macropores, assuming gravity flow
Surface runoff/ erosion	-	Based on SCS curve no. method. Erosion calc. using over*land, channel, and* impoundment elements and soil particle characteristics	Runoff based on infiltr. surplus. Erosion not considered	Runoff based on SCS curve no. method. Erosion not considered	Runoff based on SCS curve no. method. Erosion based on USLE	Runoff based on infiltr. surplus. Erosion not considered	-	Runoff based on infiltr. surplus. Erosion not considered
Macropores	-	-	-	-	-	Included, source of flow is overland surplus	-	Included, source of flow is infiltration surplus from macropores
UZ: division	Multilayer	Multilayer, but only till the end of rootzone.	Multilayer	3 layers: - rootzone - 2 deeper layers	Multilayer	Multilayer	Multilayer	Multilayer
Lower boundary	Free drainage	Free drainage	Free drainage, constant potential, lysimeter or zero flux	Horizontal groundwater is simulated	Free drainage	Free drainage or constant potential	Constant potential	Free drainage, constant potential, lysimeter or zero flux
Solute transport	Piston displacement of solute	Convective transport of solute using water flow between soil layers. Solute can move upward by capillary flow	Solves the convection-dispersion transport eqn.	Piston displacement of solute	Convective transport of solute based on water flow between soil layers	Piston displacement and mixing. Immobile and mobile *water assumed in* UZ	Solves the convection-dispersion transport eqn.	Solves the convection-dispersion eqn.
Dispersion	Tracks a non-dispersive solute point	Numerical dispersion from conv. transp. eqn. used to simulate actual solute dispersion	Calculates hydrodynamic dispersion	Calculates dispersion around midpoint of solute band using an error function	Numerical dispersion, from conv. transp. eqn. used to simulate actual dispersion	Dispersion from numerical dispersion + mobile/immobile water	Calculates hydrodynamic dispersion	Calculates hydrodynamic dispersion

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Contents	CMLS (USA)	GLEAMS (USA)	LEACHM (USA)	MOUSE (USA)	PRZM (USA)	RZWQM (USA)	PESTLA (HOLLAND)	MACRO (SWEDEN)
Sorption	One type of site only. Linear equil. sorption	One type of site only, but up to 10 solutes/metab. Linear equil. sorption	One type of site only. Linear equil. or Freundlich sorption for several compounds	One type of site only. Linear equil. sorption	One type of site only. Linear equil. sorption	Two types of sites. Linear equil. + kinetic sorption	Three types of sites. Linear or Freundlich equil. + kinetic sorption	Four types of sites. Linear equil. sorption
Input	Either K_{oc} and C-content /soil incr. or K_d /soil incr.	K_{oc} per compound and OM /soil incr.	K_{oc} per compound and org.C /soil incr.	K_d for solute in rootzone and region below.	K_d /soil incr.	K_{oc} and sorption rates per compound per site and OM	K_d and sorption rate/ soil incr. + Freundlich exp. per site	K_d /soil incr. and fraction of sorption sites in macropores
Degradation	One degr. reaction	Up to 10 degr. reactions, dep. on T, θ, and pH	2 degr. reactions per compound, dep. on T	One degr. reaction	One degr. reaction	More degr. reactions, dep. on T and θ	One degr. reaction, dep. on T, θ and soil depth	Four degr. reactions (liquid/solid, matrix/mp's), dep. on T and θ
Input	Solute half life /soil incr.	Transformation rate per compound /soil incr.	2 degr. rates per compound/soil incr.	Solute half life /soil incr.	Degr. rate coeff. / soil incr.	Solute half life per compound	One degr. rate coeff.	Four degr. rate coeff. /soil incr.
Differentiated degradation	-	-	(+)	-	-	Includes biodegradation, oxidation, complexation, photolysis, hydrolysis	-	-
Volatilization	-	+	+	-	-	+	-	(-)
Plant uptake	-	+	+	-	+	+	+	+
Evapotranspiration	Daily PET. Water removed from wettest layer first	PET calc. from R and T. AET calc. from PET, LAI and θ	AET calc. from weekly PET, root distr., resistance and θ	Daily PET. AET calc. from θ and ET input parameters	Daily PET. AET calc. from ET coeff., θ and root distr.	Calculated by growth model	Daily E_p. AET calc. from ET coeff., LAI, rooting depth and θ	Daily PET. AET calc. from PET, root distr. and θ. Water uptake preferably from mp's
Roots	Max. root depth, const. root biomass, uniform root distr.	Max. root depth, exp. decr. root biomass with depth	Root biomass const. over time or incr.	Uniform root distr., root depth can vary over time	Root biomass const. or incr., linear distr. with depth	As above	Max. root depth Uniform root distribution	Root depth const. or incr. Root length exp. distr. with depth
Other functions included			Calculation of heat flux and temp. in soil	Snow melt	Snow melt	Calculation of heat flux and temp. in soil, bicarbonate buffering, diss./prec. of $CaCO_3$, Gypsum and Al-hydroxide		Calculation of heat flux. Subsurface drainage. Swell/ shrink of mp's. Switch betw. 1 & 2 domains
Management functions	-	+	+	-	-	+	-	(+)

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2.2 Evaluation procedure

The evaluation was carried out in two steps. Initially, the physical and chemical complexity of the process descriptions was assessed on a theoretical basis. The overall approach was that a good description of water flow is a prerequisite for modelling of solute transport. The major criteria for the theoretical evaluation was therefore whether the hydrological performance of the models was able to reflect the prevailing hydrological conditions relevant for Danish soils. If the hydrological approach used in the models were judged to be adequate, model performance was tested on data sets from two different locations in Denmark.

The evaluation of the chemical part of the models could only be based on an assessment of the relevance of the process descriptions included. This was partly due to the fact that little is known about relative importance of parameters and processes describing pesticide behaviour in the soil-water environment, and partly due to the quality of the available data sets describing distribution and leaching of pesticides. It appears that some of the chemical processes in the models are described in a very simple way (e.g. linear adsorption, first order degradation kinetics), but at the same time the models seem to be ahead of the parameter generation. For instance, MACRO allows for different breakdown conditions in macropores and matrix although little data are available for parameterization.

2.3 Evaluation criteria

The two major soil types used for agricultural practice in Denmark can be described as follows: 1) Deep coarse sandy soils located in the western Jutland. These soils generally belong to the classes Inceptisols, Spodosols or Entisol and are characterized by a rather deep unsaturated zone, high amounts of recharge and hence no artificial drainage and little macroporosity. 2) Moraine clay soils located on most of Sealand and in the eastern parts of Jutland. These are generally Alfisols, but Cambisols and Ultisols occur. In contrast to the sandy soils, the moraine clay soils often contain groundwater at shallow depths, are artificially drained and have less recharge. Additionally macropores have been identified in both top soil /36/ and in the saturated zone /22/ of many of these soils.

For studies of pesticide leaching the depth and variation in groundwater level and the depth of the unsaturated zone are very important factors. In areas with deep unsaturated zones more time is available for dissipation mechanisms of the pesticides, whereas in areas with shallow and fluctuating groundwater, the pesticide may reach the groundwater very quickly and more or less untransformed. Models aiming to predict pesticide leaching under these conditions must be able to handle these fluctuations in a proper way. This implies that the hydrological part of the models should be able to handle upward flow, artificial drainage as well as fluctuating groundwater tables.

As pesticide degradation, among other factors, strongly depends on temperature, the models should also contain a description of temperature and heat flux through the soil profile.

2.4 Data

Field data from two catchments typical for Danish conditions were provided by the National Environmental Research Institute /16/. Højvads Rende (LOOP1) represents the loamy soils (mostly moraine soils) in the eastern parts of Denmark. The yearly average rainfall and recharge in this area is 550-600 mm and 100-200 mm, respectively. The area is artificially drained with tile drains located between 1.1 and 1.6 m below the soil surface. The groundwater level varies over the year between 0.8 and 1.6 m. Bolbro Brook (LOOP6) represents the sandy soils in the western part of the country. The yearly average rainfall and recharge in this area is 750-800 mm and 300-400 mm, respectively. The groundwater levels varies between 1 m and 1.8 m. Both catchments are characterized by rather flat topography.

The observation period lasted 3 years, starting in the beginning of 1989 and ending in 1991. The study was conducted concurrently with a comprehensive monitoring programme aiming to follow the variation and development of nutrient losses from agriculture throughout the country. Time series of precipitation, potential evapotranspiration and air temperature for the two catchments were provided on a daily basis. On average, the precipitation was 40 % larger in LOOP6 than in LOOP1. The methods used for collection of weather data are described in /28/.

The data provided for calibration of the water balance of the models, were time series of the groundwater levels, and additionally drain flow rates from LOOP1. The methods for installation and the location of piezometers together with the data collection scheme are described in /13,14,30/. The flow rates in the drains have been measured and data logged using 30 degree Thomsen weirs. The installation of the drainage stations on LOOP1 is described in /18/.

The soil physical and chemical data originate from analysis of soil profiles and soil samples collected in excavation pits on the plots /21/. Soil samples were taken at 4 to 6 soil depths, from 10 cm to a maximum of 2 m. Bulk density, porosity and retention properties were determined on undisturbed soil cores, whereas texture and chemical properties (organic matter, carbonate content and pH) were determined on disturbed samples. The soil properties representing the two catchments are shown in Tables 2 and 3. The clay content in the loamy soil is fairly constant through the profile, around 13 %. In the sandy soil the total sand fraction is between 70 % and 95 % and increasing with depth. The organic matter content of the soils is 1.7-1.8 % in the top soil of the loamy soil, and up to 9 % in the sandy soil and is decreasing with depth. pH varies from 5.2 to 6.6 on the sandy soil and from 7.2 to 8.4 on the loamy soil. The loamy soil becomes more alkaline with depth as the $CaCO_3$-content increases.

Neither soil moisture, tensiometer measurements or saturated hydraulic conductivity were provided for the unsaturated zone. This means that a detailed calibration of the flow above the drains or groundwater level was not possible.

Table 2: Soil physical and chemical properties for the loamy soil (LOOP1). ρ_b = bulk density (g/cm^3), θ_s = porosity (%).

Depth (cm)	w/w (%)								
	Clay	Silt	Sand	Gravel	ρ_b	θ_s	Org. matter	$CaCO_3$	pH
Size (μm)	< 2	2-63	63-2000	> 2000			w/w (%)		H_2O
10-20	11	24	63	< 10	1.62	37.8	1.7	0	7.3
32-42	10	21	68	< 10	1.71	34.3	1.8	0	7.2
50-60	12	16	71	< 10	1.77	32.5	1.0	0	7.3
90-100	18	41	39	< 10	1.96	25.9	0.2	0	7.5
120-130	13	25	53	< 10	1.77	33.5	0.1	9.0	8.3
190-200	12	23	46	< 10	-	-	0.1	19.8	8.4

Table 3: Soil physical and chemical properties for the sandy soil (LOOP6). ρ_b = bulk density (g/cm^3), θ_s = porosity (%).

Depth (cm)	w/w (%)								
	Clay	Silt	Sand	Gravel	ρ_b	θ_s	Org. matter	$CaCO_3$	pH
Size (μm)	< 2	2-63	63-2000	> 2000			w/w (%)		H_2O
10-20	5	13	76	< 10	1.30	47.7	6.4	0	6.6
30-40	5	7	79	< 10	1.13	53.5	8.7	0	6.3
45-55	4	6	86	< 10	1.26	50.4	4.2	0	5.7
65-75	3	2	93	< 10	1.32	49.6	1.1	0	5.2
100-110	3	3	94	< 10	1.53	42.0	0.4	0	5.2

3. RESULTS

3.1 Model Comparison

Stipulating the criterion that the pesticide part of the models can only be properly evaluated if the water part of the model is applicable and can be calibrated to match the hydrological conditions observed, the hydrological abilities of the models is the primary concern of the present paper. The model comparison presented therefore concentrate on processes describing the components of the water balance, on which the selection criterias were based. For a more detailed comparison of the process descriptions used in the respective models, including the chemical aspects, see /32/.

Water Flow. Basically, two approaches are taken in the models to describe water flow through the soil matrix. The first approach is the physically based description where water flow is determined by hydraulic gradients and a hydraulic conductivity of the soil, combined with a water conservation consideration in the Richard's equation. This approach requires input data of the hydraulic conductivity function $K(\psi)$ and the water retention function $\theta(\psi)$. Solving the equation numerically involves iterative matrix solvers. This approach is used by LEACHM, RZWQM, PESTLA and MACRO.

The second approach, of which there are different variations, is simpler and requires less input data and computation time. In principle, it is a "tipping bucket" method by which a continuous water balance for each computational layer is based upon its water storage capacity and a drainage rule. When water infiltrates, it is redistributed so that individual layers are filled to the maximum capacity (after evapotranspiration is accounted for) and excess water is allowed to percolate into the next layer. Usually the water content at field capacity and permanent wilting point is used as the upper and lower water content limits, respectively, and redistribution is assumed to be instantaneous (within one time step). This approach is used in CMLS and PRZM. However, PRZM provides an option for time dependent redistribution. The exact assumption used in GLEAMS is not clear from the literature in hand. Except for GLEAMS, none of the other models allow for upward flow, which is automatically accounted for in the more physically based approach using Richard's equation.

The tipping bucket description applied to water flow in sandy soils represents a more reasonable approximation of basic processes than when applied to fine-textured soils. Basically, the simplified water descriptions lack the physical basis and hence pose a major limitation in the practical use of the respective models.

The MOUSE model uses an approach, that can be regarded as an intermediate between the two afore mentioned, in which the drainage rate is taken into account by using the $K(\theta)$ relationship and a unit hydraulic gradient is assumed. This approach also precludes upward flow.

Evapotranspiration. The process of evapotranspiration is included in all of the models, with the degree of complexity generally reflecting the level of the other model components. The simplest approach is used in CLMS, where water is removed from each layer of the root zone proportionally to the amount of available water. A constant, uniform root distribution is assumed. The other models all allow the root depth to vary with time, making seasonal and yearly simulations possible. Potential evapotranspiration is input data in all the model, except in GLEAMS, where solar radiation, temperature and latitude is used to calculate potential evapotranspiration. The method used to estimate the actual evapotranspiration rate varies between the models, but generally includes some consideration of available water and the root distribution and/or the leaf area index. In MACRO, the evapotranspiration is preferentially satisfied from water in the macropores. In RZWQM, the source is not known.

Macropores. Macropores or the phenomenon of preferential flow are considered in RZWQM and MACRO only. The descriptions are similar as the flow domain is divided into two regions with specific characteristics and exchange which allows the passage of water and solute between them. Flow into the macropores stems from excess infiltration into the matrix. In RZWQM the flow inside the macropores is solved by Poisseuille's law, assuming gravity flow in full-flowing cylindrical or planar pores, which means that the dimensions of the macropores must be given. In MACRO, Darcy's law is solved for a unit hydraulic gradient with a water content dependent hydraulic conductivity. Macroporosity must be specified for each layer, and macropore geometry is only indirectly accounted for by introducing a tortuosity factor. Macropores can be continuous or discontinuous in both models. The transport from macropores into the soil matrix is incorporated as sink terms and driven by capillarity (Green & Ampt approach) in RZWQM, and by water content gradients in MACRO. Since little is known about the exact mechanisms and the quantities involved, either method seems applicable. Apparently only MACRO allows for flow in the reverse direction, from matrix to macropores.

Drainage. Only MACRO offers an option to include subsurface drainage, which is relevant under Danish conditions.

Based on the initial model comparison, it appeared that LEACHM, RZWQM, PESTLA and MACRO were the best candidates for further testing on measured data. These models all have a physically based description of water transport. The solute transport in the soil is also founded on physics except for RZWQM. The other models were considered too simple in the approach to water/solute transport. Possibly they could be calibrated to work for sandy soil conditions, but transfer of parameters between locations and interpretation of the model results is expected to be difficult.

3.2 Test of hydrological performance

RZWQM. From a study already conducted using RZWQM on Dutch data to describe nitrate movements, it was found that the model exhibited instability problems during groundwater rise. Also the model description of crop development is specific to potatoes and corn, which made application to different crops difficult /17/. These points made the model too limited for further testing in the present study.

PESTLA. It appeared that PESTLA overestimated the evapotranspiration resulting in lack of recharge and pesticide leaching. The model was therefore not tested further. Although there is no reason why the description of pesticide transport should not be adequate, it seems that the hydrological part of the model needs some modification to be able to handle the conditions observed during this study.

LEACHM. The driving variables of rain and potential evapotranspiration were required on daily and weekly basis, respectively. The daily rainfall amount was distributed evenly over the duration of the day. The soil profile depth and the number of layers of equal thickness used for the simulations had to be specified.

For the sandy soil (LOOP6), a total soil profile of 2 m was included in the model, and a constant pressure head at the lowest node was used as lower boundary condition. This implies a groundwater table that never gets below a certain value, but in periods of high percolation the water is allowed to raise above the minimum level. The deepest observed groundwater level for the two year observation period was used as input to the model (1.8 m). This setup is reasonable if permanent groundwater is present.

For the loamy soil (LOOP1), a lysimeter bottom boundary was applied. In this case the column is cut of at a certain depth (1.2 m) governed by the depth of the tile drains. Water drains from the bottom, when the bottom layer is saturated, and a zero flux is maintained as long as the profile is unsaturated. Capillary rise cannot take place over the bottom layer, and the model then do not describe groundwater at lower levels than the depth of the column. This approach most closely mimicked the conditions in a soil with artificial drainage. The outflow from the column represents both seepage and drain flow.

During the simulations, a few problems with calibration of LEACHM was encountered. Actual evapotranspiration turned out to be very small and almost insensitive to root depth and hydraulic conductivity. It appeared that this was due to a root resistance term included in the model code. Only by cancelling this term an actual evapotranspiration of comparable magnitude to what was generated by other models could be obtained. However, actual evapotranspiration is still low in some years.

Examples of simulations on the two soil types are shown in Figures 1 and 2. The simulation results showed good agreement with observations for the sandy soil in LOOP6. However, the amount of observation data to calibrate against were relatively sparse. For the loamy soil the model was unable to generate groundwater levels and drainage flow rates that were in agreement with the observations.

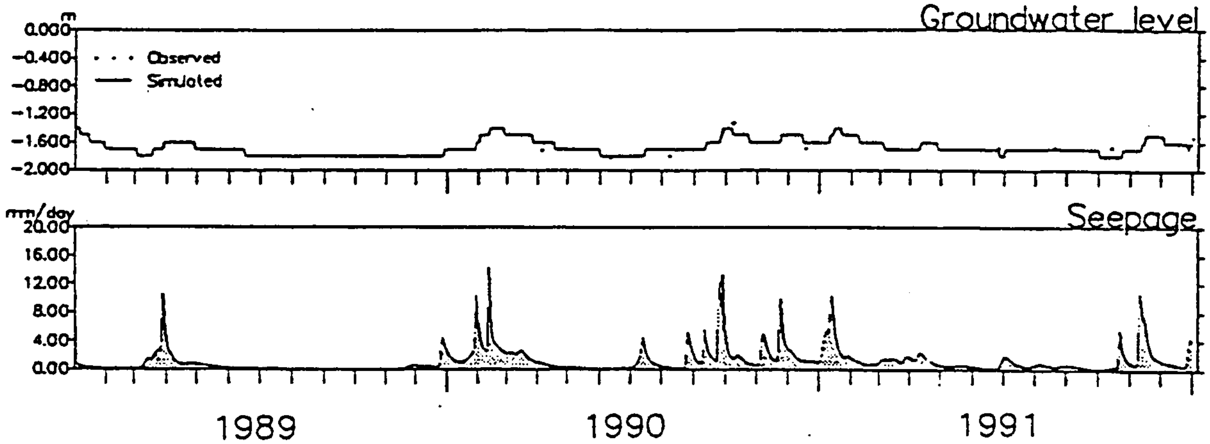

Figure 1: Groundwater level (observed and simulated) and seepage flow (simulated) on the sandy soil, using LEACHM.

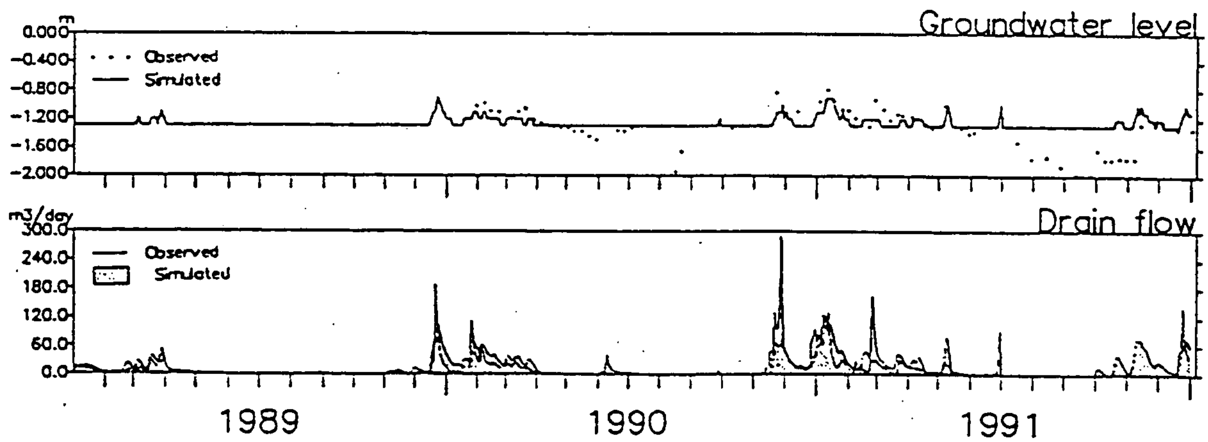

Figure 2: Groundwater level and drain flow (observed and simulated) on the loamy soil, using LEACHM.

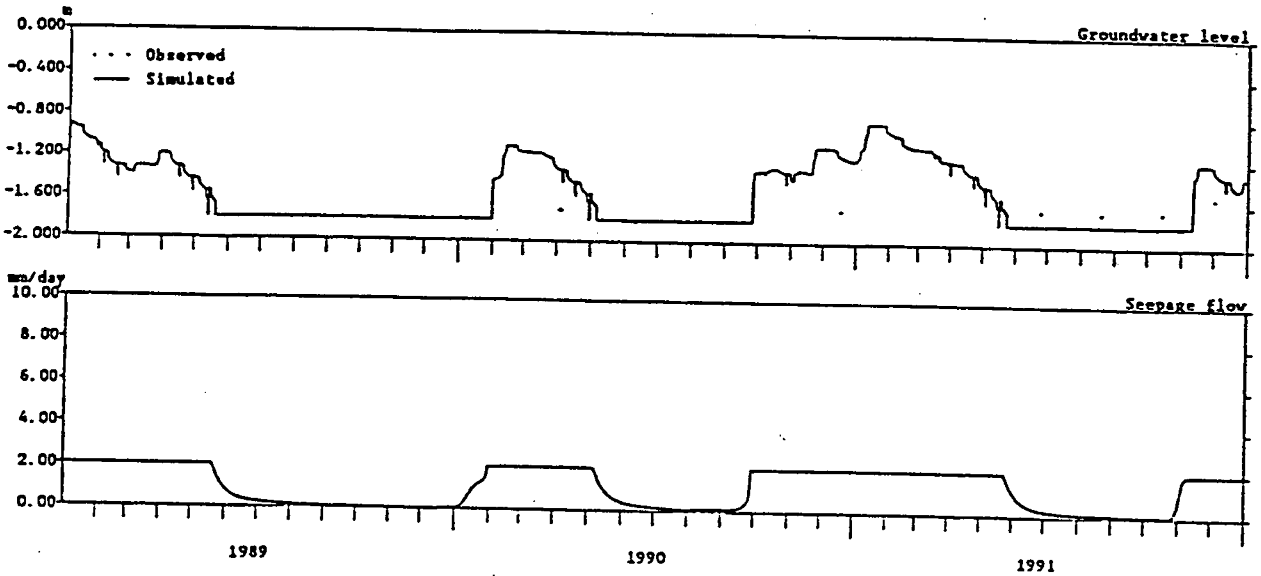

Figure 3: Groundwater level (observed and simulated) and seepage flow (simulated) on the sandy soil, using MACRO.

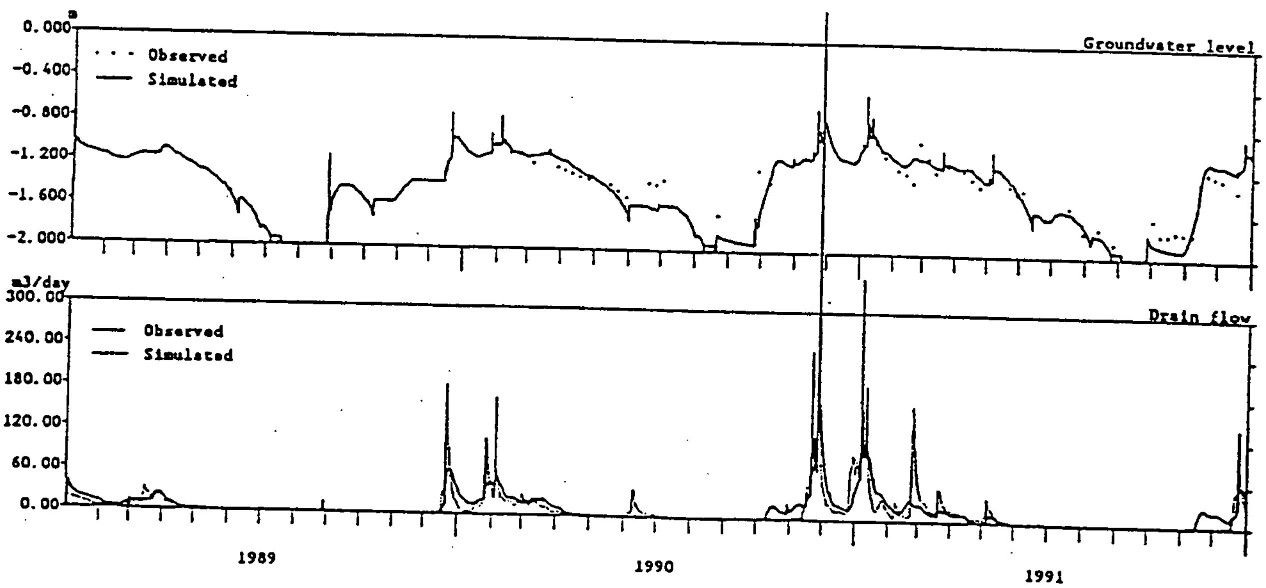

Figure 4: Groundwater level and drain flow (observed and simulated) on the loamy soil, using MACRO.

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MACRO. MACRO accepts rain and potential evapotranspiration data with a finer time resolution than LEACHM, hourly for rain and daily for evapotranspiration. However, rain still had to be input on a daily basis, since hourly data were not available. A constant rain intensity of 5 mm/hr was assumed, implying that the daily rainfall amount is distributed over a calculated duration, starting at the beginning of the day. Since MACRO accounts for macropore flow it was found appropriate to pay more attention to the rainfall intensity, than was done with LEACHM, in which the daily rainfall was distributed over the whole day.

On the loamy soil (LOOP1), the option for including horizontal flow to a tile drain was used. The depths of the drains were, as in LEACHM, assumed to be 1.2 m and the total soil profile was set to 2 m. On the sandy soil (LOOP6), the total profile depth was set to 1.8 m. For both plots, the bottom boundary condition was a zero hydraulic potential. The soil profiles were divided into 15 layers, which is the maximum allowed in MACRO. Thereby soil layers only slightly thicker than in LEACHM were used. The possibility in MACRO of accounting for water and solute transport in macropores was employed on LOOP1. For LOOP6 it was assumed that macropore effects on the solute transport would be insignificant, and only the traditional one-domain approach was used.

Examples of MACRO simulations on the two locations are shown in Figures 3 and 4. The simulations showed good agreement with observations for the loamy, macroporous and drained soil. For the sandy soil, the proper bottom boundary conditions could not be specified, and poor simulations were obtained.

3.3 Pesticide simulations

The data sets available for evaluation of the pesticide part of the models appeared to be insufficient in order to make a proper evaluation. This was mainly due to inadequate timing and frequency of the sampling programme. It could only be evaluated whether certain pesticide parameters were within a reasonable range. Both LEACHM and MACRO fulfilled this criteria.

4. DISCUSSION

Models used for pesticide registration purposes should incorporate appropriate physical and chemical process descriptions known to have significant effect on pesticide behaviour on relevant locations. For models to be used under Danish conditions this implies an ability to handle groundwater fluctuations in time, artificial drainage and preferential flow. At the time of testing one single model could not describe the range of field conditions experienced. Certain functions were lacking, such as macropores, tile drains, specific boundary conditions, or certain functions were not quite adequate (e.g. evapotranspiration calculations).

However, since the finalization of this project, some of the models have been modified in order to overcome some of the problems identified. PRZM has been modified to include algorithms based on Darcy's law in a new version named RUSTIC /35/. PESTLA has been released in a new version (2.3), which uses a more comprehensive hydrological description

/6/. Though, the model still does not account for preferential flow. In the latest version of MACRO (3.1) the problems identified regarding the lower boundary conditions have been solved, and more options added /20/. And finally, the water part of LEACHM has been modified to account for macropore flow /9/. Clearly, there is a continuant need for model comparisons, in order to asses the state-of the art regarding model capabilities under different conditions.

In the debate concerning use of models in the pesticide registration it has been argued that preferential flow should be disregarded as it is not a pesticide related property, and as the hydrological effect of macropores at the present only can be quantified through calibration /23/. However, it has been shown that preferential flow has a significant effect on the time and amount of pesticide leaching /3/. Therefore this factor should be represented in the models. Clearly the regulatory authorities cannot be expected to trust simulation results obtained with models that do not reflect important soil properties. Instead of disregarding the problem, more effort should be put into the development of tools for quantifying the effect of preferential flow in different soil types.

Another important aspect related to the use of models for registration, is the evaluation of simulation output in relation to the critical concentration limit set for drinking water. This is a question of how to determine the maximum critical concentration leaching from the root zone on certain soils, at certain depths and at certain times that will be crucial for the drinking water reaching consumers. Over which period and at what depth must the simulated concentration not exceed the limit?

The one-dimensional models considered in this project can only estimate the amount of solute leaching from the bottom of a vertical soil column representing the upper 1-2 m of the soil. However, groundwater formation and transport processes is occurring due to infiltration into deeper layers or to horizontal transport depending of the hydrogeological properties. During these processes adsorption and dilution can take place. These processes must be considered when evaluating the simulation results, and clear guidelines for assessment of the long term risk of critical pesticide concentrations reaching the drinking water must be developed. The choice of guidelines will strongly effect the number of substances being accepted or rejected.

In Holland, the threshold value of 0.1 μg/l has been adopted for groundwater at a depth of 10 m below the surface. The transport from 1 to 10 m depth is assumed to occur at a rate of 1 m pr. year corresponding to a precipitation surplus of approximately 300 mm. Due to these assumptions a transport time of 4 years has been agreed upon to be on the safe side. Additionally, first order kinetics is assumed for transformation taking place in the saturated zone.

This approach seems quite simplistic with regard to processes taking place in the saturated zone, but at least some considerations have been made in order to assess the long term responses of the hydrological system. More effort needs to be put into further investigations of transport and transformation processes taking place in the deeper saturated layers. A way of obtaining more realistic assessments of the amount of solute reaching consumers after a longer period, would be to use a physically based model describing transport and degradation processes in both the unsaturated and saturated zone, taking into account the horizontal aspect of water flow occurring in the saturated zone. At the present no model codes exist which are able to take the three-dimensional aspect of pesticide movements into account, or which are able to follow the pesticides from the root zone to groundwater aquifers or surface waters. To obtain such estimates a distributed model, such as the hydrological model MIKE SHE /1/, describing the dynamics of the entire hydrological cycle could be a valuable tool. This would make it possible to compare simulations and validations carried out on small catchments with column simulations carried out for risk analyses.

5. CONCLUSION

At the time of testing no single model was able to cover the range of hydrological conditions typical for Danish soils. In the present study, MACRO performed best on the loamy tile drained soils with macropores and varying groundwater tables, while LEACHM performed best on the more homogenous sandy soil without artificial drainage. None of the other models could be recommended. The evaluation of model ability with respect to pesticide transport was inconclusive based on the present available data material.

With respect to pesticide process descriptions, models seem to be ahead of the parameter generation. For most pesticides, little is known of the dependency of sorption and/or degradation on e.g. pH, moisture/oxygen content and depth, and although e.g. MACRO allows for different breakdown conditions in macropores and matrix, little data are available for parameterization. Furthermore, dependency of sorption on other matters than organic carbon is not well described. In order to improve these basic aspects of pesticide modelling, more process research is needed.

Additionally, comprehensive and detailed data sets reflecting pesticide transport and transformation under a diverse range of hydrological conditions are lacking, in order to perform proper model validations, and hereby to asses the possibilities and limitations of both existing models and models being improved in the future. If numerical models are to be successfully used as legislative tools, it is crucial that the simulation results are shown to be reliable. This implies that much effort must be put into development of a systematic and widely accepted validation procedure building on a representative range of comprehensive data sets.

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HOW TO HANDLE DATA FOR AND RESULTS FROM COMPLEX HYDROLOGICAL MODELS

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ABSTRACT

Numerical models for description of hydrological processes become more and more complex. At the same time the amount of data for the models and results from simulations with the models increase dramatically. We are not longer talking about one-dimensional, stationary models for water flow but three-dimensional, transient models for water flow, solute transport and reactive processes.

New presentation techniques are required in order to evaluate data and results from these models. This has implemented the development of e.g. the possibility of making "animation" of results from the MIKE SHE modelling system. Animations give the modellers the possibility to quickly inspect results from long simulations and thereby evaluate the modelling results. At the same time animations will answer many questions from "end-users" and directly be used as a result of a consultancy project.

INTRODUCTION

The amount of data for and simulation results from hydrological models is increasing as the models become more and more complex. Data and results which e.g. can include three-dimensional geological interpretations, two-dimensional potential heads or even three-dimensional, time-varying solute concentrations can not be presented properly by standard presentation techniques.

The demands on presentation technique depends on the number of dimensions in the system and the purpose as well as the "audience" for the presentation:

* **one-dimensional** data as discharge in a river at a certain gauging station is well presented in a "simple" plot with a time axis and a discharge axis;

* **calibration** of time-varying data e.g. potential head at certain observation wells is also most conveniently done by presenting data in "simple" time versus head plots;

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* presentation of **two-dimensional** data as potential head in an aquifer requires a little more, i.e. the possibility of superimposing overlay of different kind such as well location, major streets, administrative demarcations etc. Choice between simple contours (which is of interest during the calibration), contours with colours and perspective plots with different view points is a must when dealing with two-dimensional data presentation;

* presentation of **three-dimensional** data such as a pollution plume in a complex aquifer system at a certain time requires a tool where it is possible to make cuts in the system and present data as described for two-dimensional data and "exploded" plots with different view points;

* presentation of **four-dimensional** data where the fourth dimension is time (or two- and three-dimensional data where the second or third dimension is time) requires a tool which has all the above described possibilities plus the ability to make a series of plots which shows the time-varying nature of the data. This technique is called animation and has been applied in other connections since cartoons where developed but only recently in hydrological modelling.

In order to meet these demands for advanced presentation techniques the MIKE SHE modelling system has been equipped with a post-processor that on one hand is able to produce very simple plots and on the other hand is able to make advanced animations of complex data and results. Preparation of the presentations takes place in a user-friendly environment and even animations are easily prepared.

CASES

The presentation will include animations of results from three cases, i.e. a groundwater management study for the County of Aarhus, Denmark, a groundwater contamination and reclamation study for the County of Southern Jutland, Denmark, and a study of the environmental impact from establishment of a dam in river Danube, Slovakia.

Groundwater Management, Aarhus, Denmark

The municipality of Aarhus with its 250.000 inhabitants consumes approximately 25 mill. m^3/year of groundwater. During the past two to three decades this exploitation of the aquifers has lead to a decline in water table and a deterioration of flow conditions in the streams in the area. A MIKE SHE model covering 730 km^2 was developed. All relevant hydrological processes were included in the model as an integrated, transient description of:

- rainfall and evaporation/evapotranspiration processes;
- surface run-off;
- stream flow run-off;
- infiltration to upper and lower aquifers;
- groundwater flow in the different aquifers under the influence of abstraction from these;
- interactions between aquifers and streams;

One of the results from the model was water balance calculations for the deep aquifer on different topographical catchments. Fig. 1 shows results from the Giber catchment - a 46 km^2 catchment from which about 4 mill. m^3/year groundwater is abstracted since 1967 - from the period 1961 to 1990. Initially, the infiltration (45 mm/year) is in balance with an abstraction of (5 mm/year) + base-flow (30 mm/year) + subsurface flow **out** of the catchment (10 mm/year). In 1990 a new balance has appeared where infiltration (72 mm/year) is in balance with an abstraction (95 mm/year) + base-flow (2 mm/year) + subsurface flow **into** the catchment (25 mm/year).

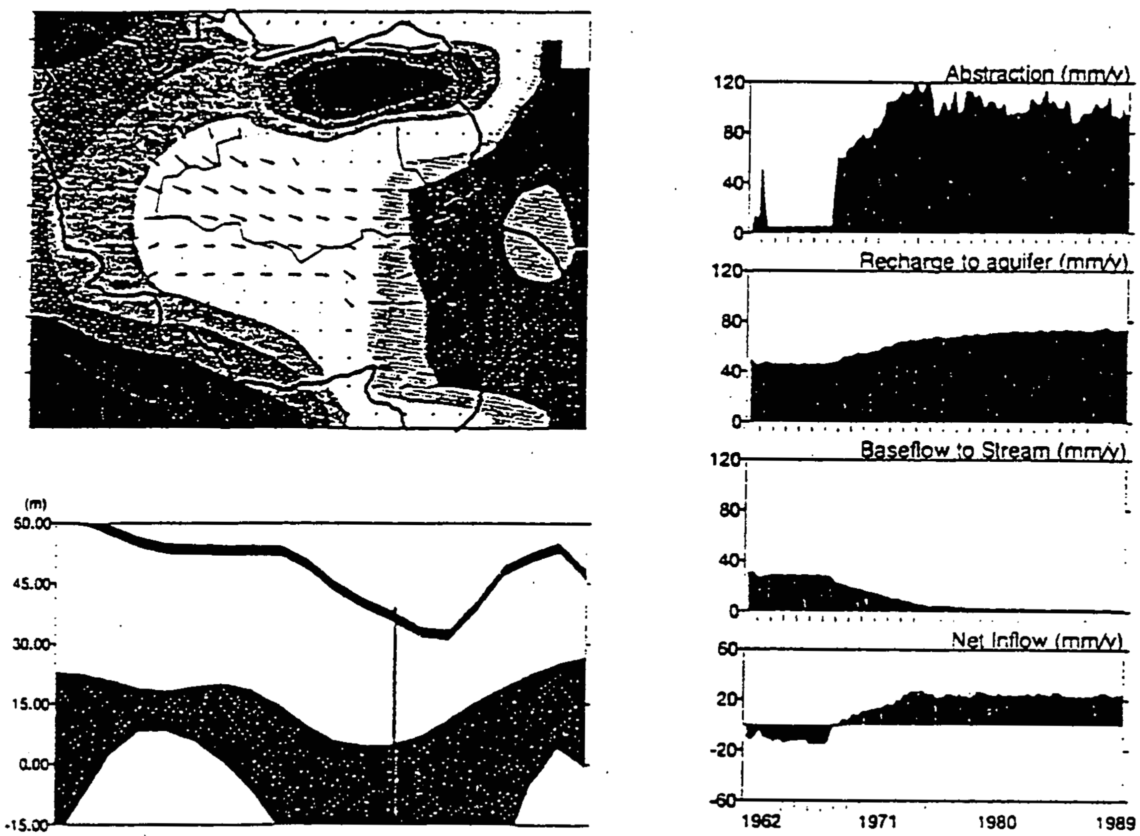

Figure 1 Water balance components for the lower aquifer in the Giber catchment.

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Animation of results from this catchment shows the dynamic behaviour of the potential head and flow patterns in the lower aquifer i.e. a generally decreasing level and a turn in flow direction in some parts of the catchment.

Groundwater Contamination and Reclamation, Southern Jutland, Denmark

A chemical waste dump located in the County of Southern Jutland, Denmark, has during a period of more than 10 years from 1964 received more than 120 tons of chemical waste primarily from a refrigerator factory. The waste includes a large amount of chlorinated hydrocarbons which have been leaching to the unconfined groundwater aquifer below the waste dump.

The waste dump was removed in 1987 and remediation wells have abstracted polluted groundwater which has been treated in an on-site sewage plant since 1988. Initially the treated water was re-infiltrated in the old waste dump to accelerate the cleaning process but this action caused an unintended vertical spreading of the plume and it was stopped in 1989.

In order to re-evaluate the remediation scheme a MIKE SHE model was developed for the area down gradient the waste dump including the aquifer and a nearby river system. The aquifer was discretized in a very detailed grid with several computational layers in the vertical direction. Calibration of the flow model against potential heads both under stationary pre-remediation conditions and during the early stages of the remediation period as well as against discharge measurements in the nearby stream gave a good description of the flow conditions in the aquifer.

Pollution patterns simulated under several alternative remediation schemes showed the a continuation of the remedial measures would have no significant influence on the future pollution of the aquifer and recipients. Steps towards closing down pumping wells and the treatment plant have been taken in order to save the operational costs which are app. 1.000.000 dkr/year.

Fig. 2 shows the pollution patterns in 1991 where the pollution almost has reached a down stream located water works and the recipient - a small stream.

Animation of results from simulations shows the development of the groundwater contamination during the period from 1964 to date and illustrates the conditions in the aquifer if no remedial actions are taken in the future.

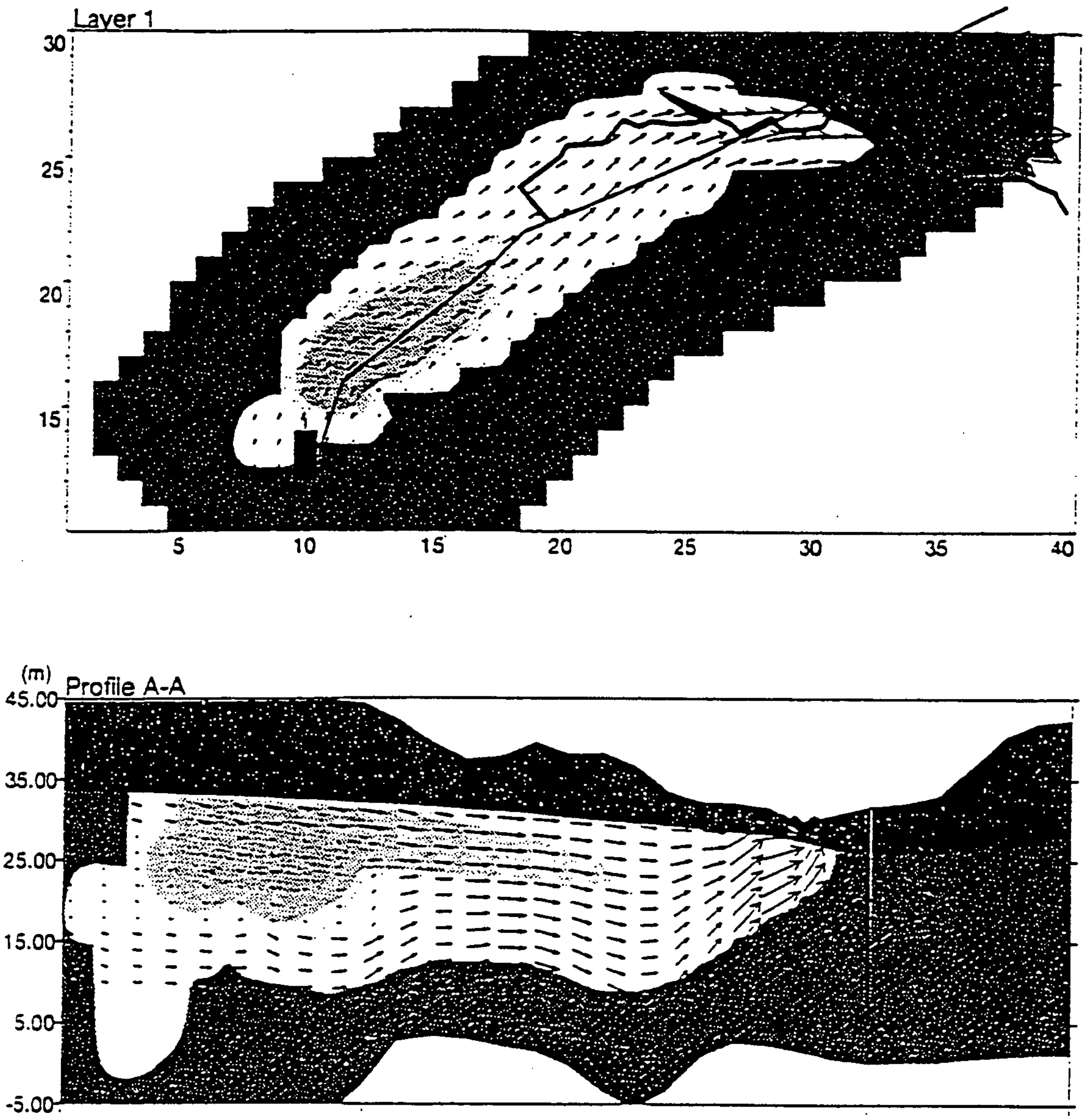

Figure 2 Groundwater contamination in a horizontal and vertical plane downstream Skrydstrup waste dump.

Environmental Impact, River Danube, Slovakia

The Danubian Lowland between Bratislava and Komárno is an inland delta formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which throughout the year receives in the order of 15 m^3/s infiltration water from the Danube in the upper parts of the area and returns it into the Danube and the drainage channels in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

The construction of the hydraulic structures e.g. in connection with the hydropower plant at Gabcikovo significantly affects the hydrological regime and the ecosystem of the region i.e. the conditions on the ground water regime as well as the sensitive riverside forests downstream of Bratislava. In spite of this basically negative trend the floodplain area with its alluvial forests and the associated ecosystems still represents a very unique landscape of outstanding importance.

To address the water resources problems in the area the project "Danubian Lowland - Ground Water Model" has been defined. An integrated modelling tool, which will form the basis for all the modelling activities, based on the Danish Hydraulic Institute's mathematical models are developed. The integrated modelling system consists of a number of models. In brief these includes **MIKE SHE** which, on catchment scale, can simulate the major flow and transport processes of the hydrological cycle, **MIKE 11**, which is a one-dimensional river modelling system, **MIKE 21**, which is a two-dimensional hydrodynamic modelling system and **DAISY** a one-dimensional root zone model for simulation of crop production, soil water dynamics, and nitrogen dynamics in crop production for various agricultural management practices and strategies. The models are all generalized tools which can be used individually or brought together in an integrated manner. Fig. 3 shows some of the various scales on which modelling are carried out.

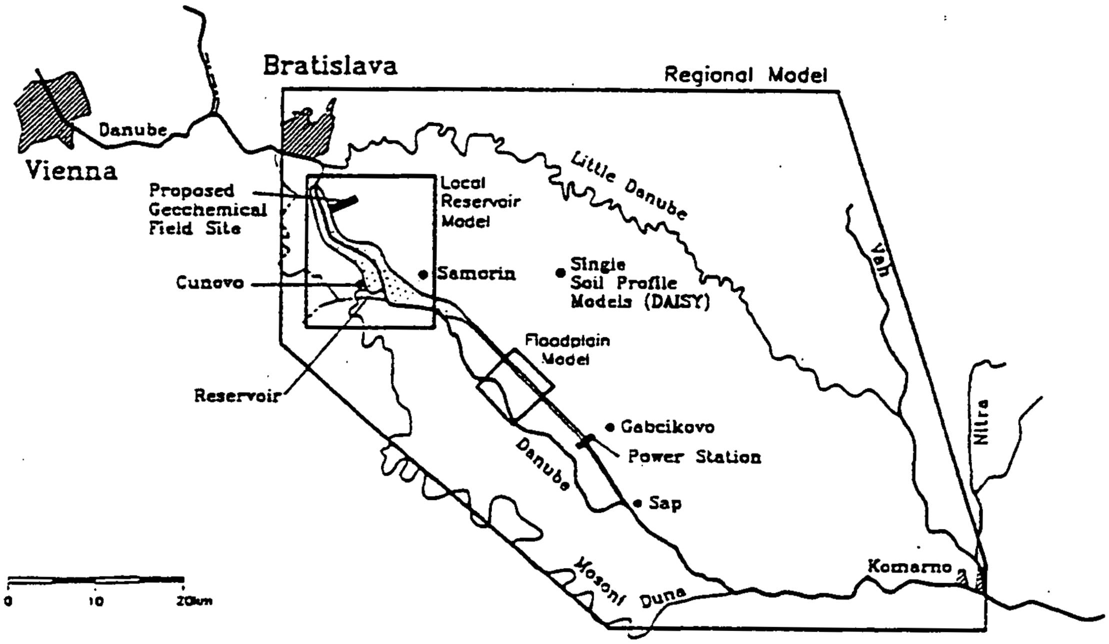

Figure 3 Illustration of the location of the area and the various modelling scales.

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Animation of results from the initial phase of the model implementation shows (1) the dynamic behaviour of the groundwater potentials dependent on the water level in the river Danube and (2) the effect on water levels in the flood plain inundations dependent on structure management practices.

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A HYDROLOGICAL MODELLING SYSTEM FOR JOINT ANALYSES OF REGIONAL GROUNDWATER RESOURCES AND LOCAL CONTAMINANT TRANSPORT

by

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ABSTRACT

Aarhus County, Denmark, is presently developing a water management plan. Increasing exploitation of the aquifers has during the past two or three decades caused an unacceptable diminishing of the stream flows in some areas and a deterioration of the water quality in othe areas. At the same time pollution from several waste sites in the vicinity of some of the major wel fields are threatening the groundwater resource. In order to address these problems an integrate hydrological modelling system which includes a three-dimensional groundwater flow and transpor model and models for the description of the processes in the upper soil part and overland part o the hydrological regime was used.

Calculations of water balances and water table variations under present and future conditions wa used to identify optimal exploitation strategies. The analyses concludes that the stream flows and groundwater quality will decrease and be further deteriorated if the present exploitation continues A redistribution of the exploitation within the existing well fields have been proposed which wil improve the conditions with regard to the stream flows.

Remedial actions have been initiated at some locations to prevent further spreading of pollution With the modelling tool one of these is re-evaluated to assess the effect of it under differen interpretations of the sub-surface geology. In this case the remediation scheme seemed to b effective also under other probable geological conditions.

The present paper describes briefly the hydrological modelling system and the its application in th analyses of ground water management in the county of Aarhus, Denmark.

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INTRODUCTION

Groundwater plays an increasingly important role in water resources in many parts of the world Traditionally, the challenges faced by groundwater hydrologists have been associated primarily t assessment of groundwater resources, which often have been considered more or less isolated fron the remaining part of the hydrological cycle. With the increased utilization of the natural resource over the past decades two new challenges have emerged:

* A need to describe the interaction between surface water and groundwater, e.g. fo prediction of groundwater recharge, including its sensitivity to drought periods, an prediction of the effects on streamflows and wetlands of groundwater abstraction.

* A need to describe the transport and fate of pollutants in the groundwater system As virtually all pollution originates from the soil surface a combined surfac water/groundwater description is also often important in this context.

Development of mathematical and numerical models to describe groundwater flow has bee subject of considerable study in the past decades. Numerous numerical models of variou complexity have been produced (see e.g. Van der Heide et al., 1985), and the mathematic foundation for solving the most important boundary value problems in groundwater hydrology i available (Bear and Verruijt, 1987, or other standard textbooks).

In the traditional groundwater modelling studies the surface water and the unsaturated zone hav been considered as boundaries. Usually, groundwater discharge to streams has been calculated a gradient dependent flow and recharge from the unsaturated zone has been calculated more or les independent on groundwater conditions.

So far much less attention has been devoted to development of integrated surface water/groundwat models. Weeks et al. (1974), Wardlaw (1978), Refsgaard and Hansen (1982), Rogers et al. (198 and Abbott et al. (1986b) have presented various surface water/groundwater models developed fc various purposes and valid for various hydrogeological conditions.

Description of contaminant transport in groundwater is a natural extension to groundwater modelling, and many groundwater models are today capable of handling both flow and trans_r_r see Van der Heide et al. (1988) for a review of many of the most widely applied models.

Two-dimensional modelling techniques are often sufficient when applying mathematical models t asses the groundwater potential for water supply. However, for analysis of the migration (pollution the assumption about fully mixing over the aquifer depth is usually not valid, hence three-dimensional approach is necessary. The three-dimensional character of solute transport i groundwater is supported by a number of international research studies e.g. Freyberg (1986 Garabedian et al. (1988), Kelley and Moltyaner (1987) and Jensen et al. (1992) who all emphasi that even when contaminant transport takes place in well-sorted so-called homogeneous deposits t spreading of the solutes has a distinct three-dimensional character.

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Different length scales apply to various problems. Generally, for ground water resource availability studies modelling areas in the order of tens to hundreds of square kilometres (regiona scale) are required, if not for other reasons then just in order to obtain adequate boundar conditions. For modelling of contaminant transport from non-point pollution sources the regiona scale may also often be relevant. For point sources, however, detailed transport modelling at a loca scale (in the order of square kilometres or less) is often required, whereas the basic flow modellin due to boundary conditions is required at a considerably larger scale. This difference in modellin scales poses a major practical problem because it is necessary to carry out both a regional flo modelling with a relatively coarse computational grid and a local flow and transport modelling wit a much finer grid. Obviously, it is important that the modelling at these two scales easily an consistently can be combined.

Hence, there is a growing need for models with the following capabilities:

* Fully integrated description of the surface water and groundwater systems.

* A generalized groundwater description applicable for confined/unconfined, mult layered or fully three-dimensional conditions.

* Combined flow and transport modelling for both the surface and groundwat systems with possibilities for linking to hydro-geochemical modelling.

* Possibility for easily combining regional groundwater flow modelling with local flo and transport modelling.

No modelling systems having these combined capabilities have been found in the literature. Th paper presents such a modelling system and its application to a specific groundwater manageme project.

DESCRIPTION OF THE MIKE SHE MODELLING SYSTEM

Model structure and process descriptions in the MIKE SHE

The Systéme Hydrologique Européen (SHE) is a generalized mathematical modelling syste capable of describing the entire land phase of the hydrological cycle in a given catchment. The fi name MIKE was introduced in connection with the development of a menu-driven user interfac The catchment is discretized in the horizontal plane by a grid square network which is used bo in the overland flow component and the groundwater flow component. These are linked by vertical column of nodes at each grid representing the unsaturated zone, see Fig. 1. Wat movements in the catchment are modelled by a numerical solution (finite difference) of the part

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differential equations describing the processes of overland and channel flow, unsaturated :
saturated subsurface flow, interception, evapotranspiration and snow melt.

The MIKE SHE is modular in structure implying that alternative options with different degrees o: complexity are available for most of the components. Thus, for example for surface water orientec studies a simple two-dimensional groundwater component may be used (Refsgaard et al., 1993; Jair et al. 1993; and Lohani et al., 1993), whereas for groundwater oriented studies simpler version: of the surface water components may be sufficient (e.g. Jensen et al., 1991 and Jensen et al (1992)).

A general introduction to the MIKE SHE modelling concept is provided by Abbott et al. (1986a,b and a detailed description of the more recent three-dimensional groundwater flow and transpor component is presented in Refsgaard et al. (1992). In the following a brief description of the MIKI SHE components utilized in the present study is given.

Interception/evapotranspiration component

The interception process is modelled by a modified Rutter model. On the basis of meteorologica input it calculates the interception loss, the actual water storage on the canopy and the net rainfa reaching the ground through the canopy. The evapotranspiration component is based on a modifie version of Kristensen and Jensen (1975), where the actual evapotranspiration is calculated from th potential evapotranspiration as a function of leaf area index and soil moisture availability in the roc zone.

Snow melt component

The snow melt component selected for use in this study is based on a simple degree-day approach A more sophisticated option also available in the MIKE SHE is described in Abbott et al. (1986b)

Overland and channel flow component

When the net rainfall rate exceeds the infiltration rate, ponding occurs. Overland flow may develo both by this mechanism or when the groundwater table rises to the ground surface. Overland flo is simulated in each grid square by solving the two-dimensional diffusive wave approximation c the St. Venant equations. For channel flow, the one-dimensional form of this equation is solved i a separate node system located along boundaries of the grid squares.

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Unsaturated zone component

Soil moisture distribution in the unsaturated zone is calculated by solving the one-dimensiona Richard's equation. Extraction of moisture for transpiration and soil evaporation is introduced vi: sink terms at the node points in the root zone. Infiltration rates are determined by the uppe boundary which may be either flux controlled (net rainfall) or head controlled in case of ponding The lowest node point included in the finite difference scheme depends on the phreatic surface leve and allowance is made for the unsaturated zone to disappear in cases where the phreatic surfac: rises to the ground surface.

Saturated zone component

The groundwater flow is modelled by using a modified Gauss-Seidel implicit, iterative finite difference solution of the three-dimensional non-linear Boussinesq equation. The solution ca: optionally be utilised for two-dimensional, multi-layered or fully three-dimensional aquifer systems as well as for any combination of confined/unconfined conditions (Refsgaard et al., 1992).

The upper boundary condition in the top layer is the recharge from or to the unsaturated zon which is coupled to the phreatic surface located in this layer. Interaction of the aquifer with th river system is calculated on the basis of the differences in river water levels and groundwate hydraulic heads.

Solute transport

A solute transport description has been developed for the combined overland/channel flov unsaturated and saturated zones as add on components to the flow descriptions.

The solute transport description is based on the advection-dispersion equation with the tradition: dispersion formulation, which in the groundwater system is generalized to anisotropic conditior in three dimensions. The numerical solution to the advection-dispersion equation is based on th QUICKEST method, which is an advanced, explicit finite difference formulation. This method i mass conservative, computationally efficient and has negligible numerical dispersion as compare to traditional finite difference schemes, Refsgaard et al. (1992).

Pre- and postprocessing of data and model results

The pre-processing package of MIKE SHE functions as a Geographic Information System (GIS One part of it includes tools for digitizing and for construction of thematic overlay maps of spatiall distributed data (geological, hydrogeological, hydrological etc.) from digital information whic becomes a more and more common way of distributing data. In the case study described belo

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information e.g. the topographical part was based on a digital terrain model which was ava[illegible] from the National Survey and Cadastre.

An other part of the pre-processing package enables automatic setup of input data files for MIKE SHE with a chosen grid scale which in principle is independent of the selected information scale Processing of geological information from the National Geological Surveys into a three-dimensiona hydrogeological model is divided into a geological part and a numerical part which in principle are independent of each other. The geological model is usually established on basis of a number o transects across the model area which provides a series of continuous x-z pictures (cross-sections showing the geological conditions. Combined with an estimate of the horizontal extent of eac geological unit a three-dimensional interpretation is established.

Both the boundaries of the different geological layers and the horizontal extent of the geologica units can be digitized. Combined with an interpretation of the distribution of hydrogeologica properties of each geological unit a three-dimensional model with arbitrary grid spacing for th groundwater part can automatically be developed by an interpolation procedure.

The post-processing package of MIKE SHE includes advanced tools for retrieval and treatment o results as well as presentation of data and results. Water and solute balance calculations can easil be carried out for the entire area and for subareas as well as for any part of the hydrological cycl and compared to measured values e.g. discharge from a certain catchment.

Presentation of data and results is a very important part of an investigation and the post-processo has the ability to present date and results in several ways from the most simple time-series print to contour maps, colour prints, exploded prints, and animation of e.g. potential head variations o the migration of a contaminant plume. Some of the possibilities are shown in connection with th case study described below.

Selected applications

MIKE SHE has during the latest five years been applied in a large number of research an consultancy projects both in Denmark and abroad.

The list of research projects includes:

> **"Development of a European Soil Erosion model", 1992-94** for the Commission (European Communities and the Danish Agricultural and Veterinary Research Council.
>
> **"Effects of forestry drainage and clear-cutting on flood conditions", 1991-92** for tl Swedish Natural Science and Research Council.

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"Modelling of the Nitrogen and Pesticide transport and transformation on catchmen scale", 1991-94 for the Commission of European Communities and the National Agenc of Environmental Protection.

"Danish research programme on groundwater pollution from waste disposal sites" 1988-90 for the National Agency of Environmental Protection.

"Danish research programme on Nitrogen, Phosphorus and Organic matter", 1986-9 for the National Agency of Environmental Protection.

The list of consultancy projects includes:

"Danubian lowland - groundwater model", Slovakia, 1992-95.

"Tapa air base - groundwater model", Estonia, 1993.

"Environmental impact assessment of a highway construction", Denmark, 1992-93.

"Optimisation of remedial actions for protection of a groundwater resource" Denmark, 1992-94.

"River management study for the river Avon", England, 1992-94.

"Identification of a new well field for the water supply of Odense", Denmark, 1991-93

"Hydrological model for water supply planning in Aarhus", Denmark, 1998-92.

"Transfer of SHE to India", India, 1987-90.

Consultancy projects are typically finalized with a transfer of the calibrated model to the clier including a training course in the use of the model for further applications. MIKE SHE is also sol as commercial software and is now installed in more than 25 institutions in Denmark and abroac

CASE STUDY

Background

Aarhus with its 260,000 inhabitants is the second largest city in Denmark. It has a total wate demand of around 30 mill. m^3/year supplied from deep confined sand and gravel aquifers.

The very intensive water abstraction that has taken place in the Aarhus area during the past 25-3 years has in several places caused a drawdown of the groundwater level by 15-20 m. Because c this drawdown the stream flow in a number of water courses has been reduced to an unacceptab

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level. At the same time the drawdown causes changes in the chemical balance of the aqu[illegible] which may lead to a deterioration of the water quality.

Old waste disposal sites containing chemical and/or household waste have proven to be serio threats to the groundwater quality and hence to the water supply. Thus, many well fields supplyir water to the municipality of Aarhus are threatened by neighbouring waste disposals. These was disposals are being examined in these years for assessment of the extent of the present and futu groundwater pollution.

In order to achieve an improved quantitative understanding of the water resources in the Aarh area and to address the above problems a comprehensive hydrological modelling study wa implemented by use of the MIKE SHE (Aarhus County and Aarhus Waterworks, 1991). The stud was carried out in close co-operation between the county of Aarhus, which is responsible for th overall water resources planning in the region, the Water Works of Aarhus and the Danis Hydraulic Institute.

The aims of the project were as follows:

(a) To establish a hydrological model capable of calculating as well regional water balance groundwater levels and discharges as local transport of contaminants from point sources

(b) To estimate the groundwater availability and analyze whether the present abstraction rat are in accordance with a sustainable development of the water resources.

(c) To predict the impacts of alternative groundwater abstraction scenarios on the groundwat conditions as well as on the surface water conditions (stream flow, wetlands).

(d) To utilize the model in planning a monitoring programme.

(e) To predict contaminant transport from specific waste disposal sites.

(f) To study the effect of climate change on the groundwater resources.

The main conclusions on item a) through e) are reported in this paper.

<u>Hydrological regime and data availability</u>

The modelling study covers an area of approximately 800 km^2 and includes catchments for fi small rivers and several smaller streams. Fig. 2 shows the topography and the river network syste which have been digitized.

The geology of the area is rather complex and characterized by a mixture of glacial and alluv deposits. The most important aquifers are "buried valleys" comprising thick (up to 100 m quaternary sandy alluvial deposits and confined below by impermeable tertiary alluvial clay a

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above by low permeable glacial till. For the geological description data from more than 100 boreholes are available in Aarhus county's computerized database.

The upper soils are sandy loam which have been characterized by a soil water retention curve wit "field capacity" (pF = 2.0) at 28 vol% and "wilting point" (pF = 4.2) at 11 vol%. The land us within the model area is given in Table 1. The urban area is assumed to consist of 1/4 imperviou area with instantaneous runoff and no infiltration and of 3/4 grass area.

Temperature data which is used by the model for calculating the snow accumulation and -melt available for one station from 1876 to date.

The annual average precipitation varies within the range of 500 mm to 860 mm in the area wi significant variations from year to year. Daily rainfall data is available from 1880 to date. Fo estimation of daily areal precipitation in 17 subareas, data from 15 stations have been utilized.

Table 1 Land use in the 800 km^2 model area

Landuse	Areal distribution (%)
urban areas	13
forest areas	12
wetland areas	2
grass	23
winter sown serials	23
spring sown serials	27

The potential evapotranspiration is estimated from pan observations from one station. The averag potential evapotranspiration is 600 mm/year.

Data on groundwater abstraction from 77 "well fields" within the area has been compiled on monthly basis from 1930 onwards. Data on piezometric head/groundwater table exist from 9 observation wells, typically with four observations per year. The longest records date back to 19 when the groundwater abstraction was very insignificant as compared to today.

Discharge data time series exist for about 10 gauging stations. One series goes back to 1920, b the majority dates back to the 1970'ies and 1980'ies. For some catchments discharge data exis for periods both before and after the establishment of major groundwater abstractions.

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Establishment of a geological model

As a necessary basis for the preparation of input data for the groundwater part of the MIKE SHE a geological model was established. A geological model is a geometrical representation of the mos: important geological layers. A geological model is established on the basis of an understanding o: the geological formation process and the available local geological data. Hence, geologica interpretation constitutes a very significant role in the establishment of a geological model.

For the present purpose the geology within the modelling area was interpreted to compose the following five layers:

(1) An upper thin layer consisting of the upper three m of soil. This layer contains the root zone and in general the entire unsaturated zone and the upper phreatic surface. In the major part of the area this soil is drained through artificial tile drains.

(2) A moraine clay layer with low permeability. This layer functions as an aquitard.

(3) An upper sandy aquifer layer, from which a minor part of the groundwate abstraction takes place.

(4) A second moraine clay layer with low permeability. This layer functions as a: aquitard.

(5) A lower aquifer mainly composed of quaternary sand and gravel deposits from where the main part of the groundwater abstraction takes place.

The lower aquifer is confined below by tertiary clay deposits which are assumed to be impermeabl (i.e. no flow boundary).

Not all of the five layers are present all over the area. Like all models this five-layered geologica model is a simplification of the real world. Thus, e.g. the lower aquifer at some places consists o several layers with significant variations in hydraulic conductivity. However, it is believed to reflec the most significant geological features of the area.

The procedure for establishing the geological model was principally as follows:

(a) The existing maps showing e.g. elevation of the impermeable bed, thicknesses of th various aquifers and aquitards were digitized and fed into the MIKE SHE geologic pre-processor. The topographical part was based on a digital terrain model whic was available from the National Survey and Cadastre. These maps did not all cove the entire model area (yet the most important areas were covered) and hence inte: and extrapolation assumptions had to be made. All the existing information wa basically two-dimensional maps, which did not in all respects prove to be full consistent when subject to display and further analyses in the three-dimension:

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geological model context. Hence, some minor modifications were introduced. Th result of this was the first and very preliminary version of the geological model.

(b) Hydraulic parameter values were assumed for the various layers, generally by usin identical values all over the area for the respective layers. For the lower aquife: however, three different hydraulic conductivity values were used to reflec hydrogeological maps showing "good", "medium" and "poor" transmissivities. . preliminary simulation was then carried out with the hydrological model. B comparing the simulated hydraulic head pattern with the observed one majc discrepancies occurred in many parts of the area.

(c) A series of 11 North-South cross-sectional profiles covering the entire modelling are was established by use of geological and hydrogeological information from all th wells available within a distance of 1 - 2 km from the profiles. Inconsistencies in th simulation results of the hydrological model based on the preliminary geologic: model were taken into account in the geological interpretation. These geologicall interpreted profiles were then digitized and fed into the MIKE SHE geological pr processor, which through interpolation routines and other data manipulatic programs generated the second version of the three-dimensional geological model

(d) Based on the second version of the geological model the calibration of th hydrological model was initiated. The hydraulic parameter values were allowed : vary within limits, which could be justified as realistically consistent with th geological model. A significant part of the calibration was successfully complete at this stage. However, still some major inconsistencies occurred in certain area in the sense that it was not possible to obtain a satisfactory agreement betwe observed and simulated hydraulic heads.

(e) In the areas where such major inconsistencies occurred the geological model was r interpreted. In some cases new cross-sectional profiles were established with small distances than used in connection with preparation of the second geological mode This process resulted in the third geological model.

(f) The final calibration of the hydrological model was based on the third version of t geological model.

Thus, this procedure was iterative with comprehensive interaction between hydrological modelle and geologists. By using geological data directly as input to the model the model assumptions a much more transparent to non-modellers. It is our general experience that the interdisciplina cooperation between modellers and geologists becomes much more fruitful in this way.

During the interpretation of the second and third geological model and the calibration of t hydrological model the geologist in the administration of Aarhus County was forced to illustr the aquifers and aquitards in a three-dimensional picture. In this way their geological understandi has been changed in some areas and certainly strengthened. As a direct consequence of thi: groundwater resource monitoring network has been re-evaluated to obtain a better utilization of the economic resources. Several monitoring wells have been closed down and a series of new wells with a more optimal location has been proposed of Furthermore, it is possible to update the model as information from new wells and other geological investigations are collected in the future.

The resulting geological model is illustrated in Fig. 3 showing the five layers in a typical cross-section.

Setup of the regional hydrological model

Most of the spatially varying data have been prepared in a 250 x 250 m^2 squares resolution. By use of the MIKE SHE pre-processing programs additional model setups corresponding to 500 x 500 m^2 and 1000 x 1000 m^2 were then automatically generated. The main calibration work was carried ou by using the 500 x 500 m^2 net. The 1000 x 1000 m^2 net was not sufficiently fine to resolve all the variations in topography and geology but was, for saving computer time, used in some of the preliminary calculations of long time series. The 250 x 250 m^2 net was used in connection simulations for some subareas, e.g. the Giber Å area described below.

Hydrogeological information about the horizontal and vertical hydraulic conductivities as well as the specific yield and storage coefficients are necessary for setting up the subsurface part of the modelling system. The distribution of the hydraulic conductivities and storage coefficients were estimated by analysing test-pumpings and specific capacities of several hundred wells. Fig. 4 shows the transmissivity of the lower aquifers varying from less than 1 m^2/day to more than 200 m^2/day

The boundary conditions for the aquifer system have been specified as constant head. This is a good assumption for the eastern boundary, which is the sea. For the boundaries in the other direction: it may not be fully correct in all cases; however, these boundaries are located so far away from the well fields and areas of interest that uncertainties on these boundary conditions have negligible effects on the main results. The surface water system comprises several independent river systems Specific information on internal topographical divides are not required as input data since the overland flow in MIKE SHE is calculated directly on the basis of topographical data.

The model area, which covers approximately 800 km^2, was divided into 3213 computational points in the horizontal plane. The unsaturated zone was assumed to be less than 3 m and was divided into 30 calculation points. In order to save computer time the unsaturated zone flow was not calculated in each computational point i.e. in areas with identical conditions (rainfall, temperature, potential evaporation, vegetation, soil type, soil profile and depth to the phreatic surface) representative calculation of the unsaturated flow was carried out and in this way the number of representative columns was reduced to 197.

The model was run with a basis time step of 6 hours, but has automatic facilities for decreasing the time step when required for numerical reasons, e.g. in connection with rainfall events. For illustration of the computer requirements, the entire regional model in a 500 x 500 m^2 net required

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7.7 MB RAM and the computational time was approximately 1 CPU hour per year of simulatic on a HP 9000/720 (50 MIPS) computer.

Together with the geological model the calibrated hydrological model was installed at a comput at Aarhus County administration. The staff has been trained in making updates of the geologic model and running the hydrological model with e.g. different abstraction schemes. Thus, tl modelling system has now become an important tool in the daily work with permissions for wat use and other issues on groundwater management.

Application to regional water resources assessment

The entire study area comprises several smaller catchments, and in the following results from tl Giber Å catchment are shown. The Giber Å is located in the south-eastern part of the area (see Fi 2) and has a topographical catchment area of 46 km^2 at the gauging station. Well fields have be established close to the river, and the groundwater abstraction has increased from 0 to 5 mi m^3/year from 1965 to 1975. During the same period large drawdowns and significantly smaller lo flows have been observed.

These observations become obvious from calculations of the water balance of the lower aquife In Fig. 5 the simulated time series of the four water balance components of the lower aquifer shown:

* groundwater abstraction
* discharge from the subsurface systems to the river (base flow)
* recharge from the upper aquifers
* inflow across the catchment boundaries

From Fig. 5 it is seen that the abstraction which is about 100 mm/year results in the followi changes of the components in the water balance equation:

* the recharge to the lower aquifer has increased from 45 to 75 mm/year

* the initial outflow of 13 mm/year across the lower aquifer "boundaries" has turn to an inflow of 25 mm/year

* the base-flow to the stream is reduced from 30 mm/year to nearly zero

Comparison between measured and simulated discharge at the catchment outlet (see Fig. illustrates the effect of the abstraction on the river flow; during the dry summer seasons t discharges have been reduced significantly.

The effect of the abstraction on the hydraulic head in the aquifer is also evident. From figure which shows both the temporal development of the hydraulic head in selected wells and in t

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entire catchment, it is seen that the drawdown is as high as 15 m in selected wells and the o drawdown is also significant.

The figure also shows the changes in groundwater flows i.e. the flow direction in some parts o the catchment (south-west boundary) has turned almost 180° from 1963 to 1978.

The effects of various alternative future groundwater abstraction schemes have been predicted b use of the calibrated model (Aarhus County and Aarhus Waterworks, 1992). The results are show in Fig. 8 for a selected groundwater well and in Fig. 9 for the Giber Å discharge.

The selected schemes are:

- 100% : Future abstraction rates same as actual pumping in March 1986 at all we fields;

- 25% : Future abstraction rates 25% of March 1986 pumping;

- 75% : Future abstraction rates 75% of March 1986 pumping;

- 125% : Future abstraction rates 125% of March 1986 pumping;

- A : Special scheme with individual adjustments at each well field. At the two well fielc within the Giber Å catchment the future abstraction rates will be 2.0 + 0.9 mi m^3/year as compared to the 3.5 + 0.8 mill m^3/year actually pumped in March 198

The simulations have been carried out with the 1986 situation as initial conditions using the 1961-8 meteorological time series as input data for the 20 year simulation 1986-2005.

The results in Fig. 8 indicate that the groundwater resources conditions, which have bee characterized by a continuous reduction in head since the start of the abstraction, see also Fig. will be stabilized and even slightly improved by the scheme A. On the other hand a continuatic with the present abstraction rates (scheme 100%) is likely to result in marginally further reductic in head until a steady state occur.

The effect of three alternative schemes on the low flow situation is illustrated in Fig. 9. F instance, it appears that with a continuation of the present abstraction rates (100%) the discharg will be less than 50 l/s in average for 35% of the time and the minimum flow will be 18 l/s.

Application to local contaminant transport modelling

As a case study, illustrating the models capability to simulate the transport in the groundwat system of contaminants from waste disposals, results from an application which covers three was

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disposal sites (Aarhus County, 1992 and 1993) located in the middle of the model area is shown in the following.

Solute transport of pollution from joint sources is a local problem as compared to the groundwater abstraction problems discussed above. Hence, solute transport modelling has to be carried out at a much smaller scale and with a much more detailed flow description than the regional groundwater flow modelling. On the other hand a local groundwater model is very much dependent on the boundary conditions to the flow model, so that it cannot be established independently for a small area of relevance for solute transport modelling. Furthermore, in the Aarhus area the regional flow pattern is to a large extent determined by the groundwater abstraction from the various well fields, so that one of the management options in connection with groundwater pollution is to change the local flow directions by redistributing the pumping among the various well fields. Consequently, a modelling approach with a combined regional flow model and a local flow and solute transport model is required.

The basic regional flow modelling was carried out with a horizontal discretization of 500 m x 500 m^2 and a five layered vertical discretization. Based on this model, boundary conditions in terms of simulated time series of hydraulic heads etc. were generated for subsequent use by a local model.

The local model, comprising both flow and solute transport, was discretized into a 125 x 125 m horizontal net and 10 m net in the vertical direction resulting in a fully three-dimensional solute transport modelling. This model setup was used to investigate whether the solutes were likely to migrate from the waste disposal sites to the nearby well fields.

Another local model, comprising both flow and solute transport, was established in order to optimize the remedial actions for the upper part of the aquifer system. It was discretized into a 25 x 25 m^2 horizontal net and 5 m net in the vertical direction.

The location of the waste disposal sites and the model setups are shown in Fig. 10.

In connection with the hydrological modelling of the pollution from the Eskelund waste disposal sites local field investigations have been carried out. On basis of geological information from several wells in the local area the overall geological setup was updated.

As a result of the updated geological setup a much more heterogeneous aquifer/aquitard system was recognized in the area. Three aquifers were identified in the quaternary sediments - an upper unconfined, a lower confined and an intervening confined. The aquifers are separated and surrounded by low permeable layers of glacial till. However, still considerable uncertainty about the connection between the aquifers existed and the local model was applied to assess the spreading of contamination under different geological interpretations.

Three possible geological interpretations were identified and modelled in a 125 m grid resolu

> geo-setup 1: first trial only based on field investigations - almost no connection between th aquifers;
>
> geo-setup 2: most probable interpretation - some connection between the lower and th intervening aquifers;
>
> geo-setup 3: "worst case" interpretation - good connection between the lower and th intervening aquifer.

The spreading of the pollution in a cross-section through the waste sites and towards th waterworks is shown in Fig. 11 for the three different geological interpretations. The three differen geological interpretations do not result in significant differences in the hydraulic heads of the lowe aquifer. However, it is evident from Fig. 11 that thy have considerable effect on the spreading o dissolved solutes. It is seen that the pollution becomes a threat to one of the waterworks after 2 years of leaching if the "geo-setup 3" is a reality.

The simulations with the local model with the 25 m grid network show however, that the vertica extent of the pollution plume most likely will be limited to the upper part of the semi-confine aquifer and the horizontal extent of the plume will limited by the streams which apparently drai most of the pollution and prevents further spreading.

CONCLUSIONS

Integrated mathematical models describing the interaction between surface and subsurface wate systems in a physically correct way are necessary tools to analyze the effects of human activitie on the hydrological cycle.

The presented results, which constitute a very small part of the Aarhus study, have clearl demonstrated that the MIKE SHE successfully can be applied for analysing regional surface wate bodies and subsurface groundwater systems and particularly the interaction between the two. Wit the calibrated model successful studies have also been carried out on optimizing the ground abstraction with minimized adverse effects in terms of water table drawdown and stream flo depletion.

Furthermore, as the geological setup is directly based on geological profiles, it is possible to appl the model for analysing the possibilities for groundwater development at new well fields the b model simulations, as well as for evaluating different strategies for the ongoing groundwate abstraction. Through the establishment of the geological model and the calibration of th hydrological model a great knowledge about the geometry of the subsurface environment has wa obtained and stored in the database of the modelling system. Based on this a much more efficier monitoring system are being implemented. The geological interpretation is still improved throug new borings and the changes are incorporated in the hydrological model routinely by th administration of Aarhus County.

The three-dimensional modelling of contaminant transport in the groundwater due to waste dispos: sites has been illustrated by use of a combined regional model/local model approach, where time varying boundary conditions for a local model with fine discretizations are generated by a region: (flow) model with a more coarse discretization.

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Fig. 1

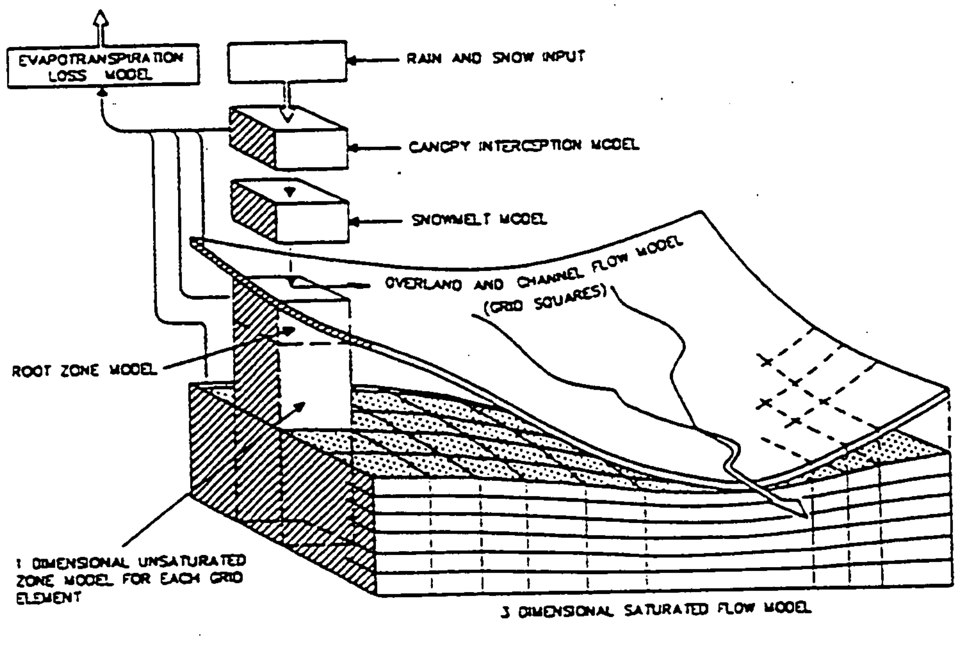
EVAPOTRANSPIRATION LOSS MODEL
RAIN AND SNOW INPUT
CANOPY INTERCEPTION MODEL
SNOWMELT MODEL
OVERLAND AND CHANNEL FLOW MODEL
(GRID SQUARES)
ROOT ZONE MODEL
1 DIMENSIONAL UNSATURATED ZONE MODEL FOR EACH GRID ELEMENT
3 DIMENSIONAL SATURATED FLOW MODEL

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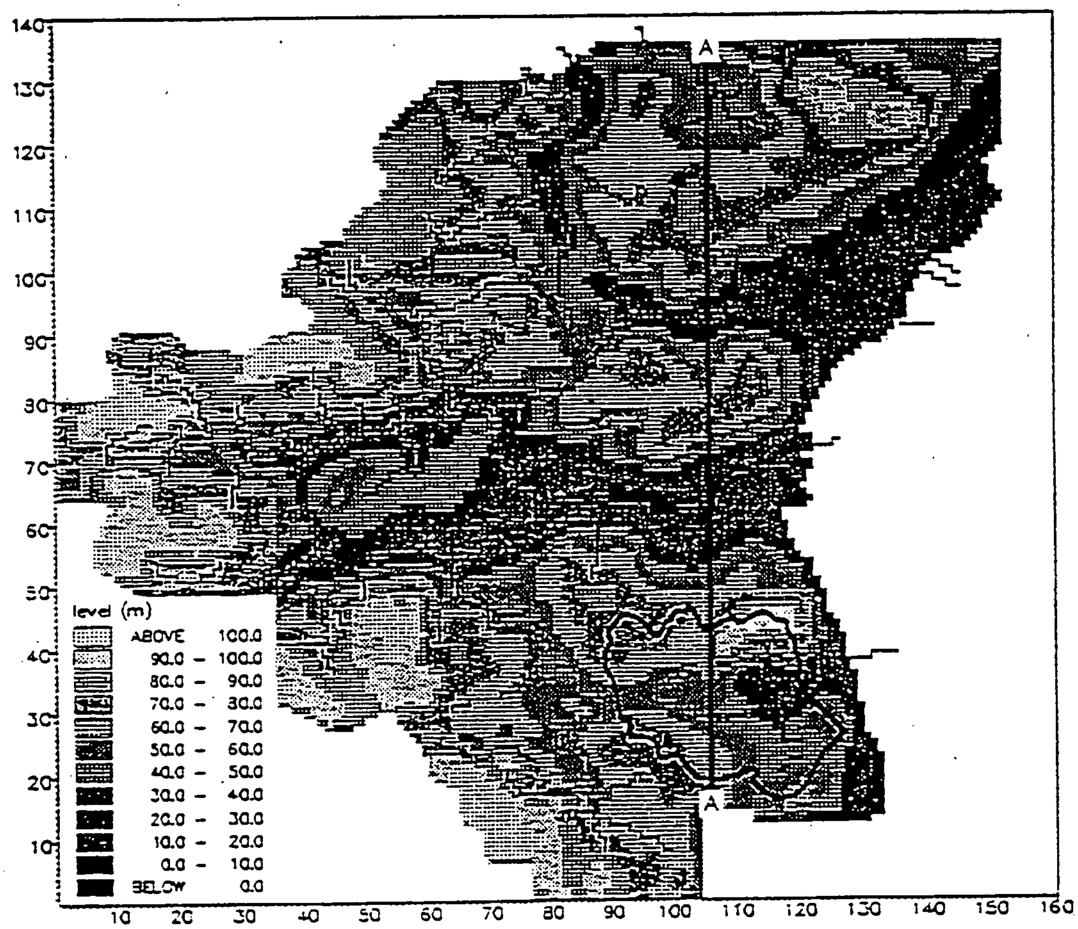
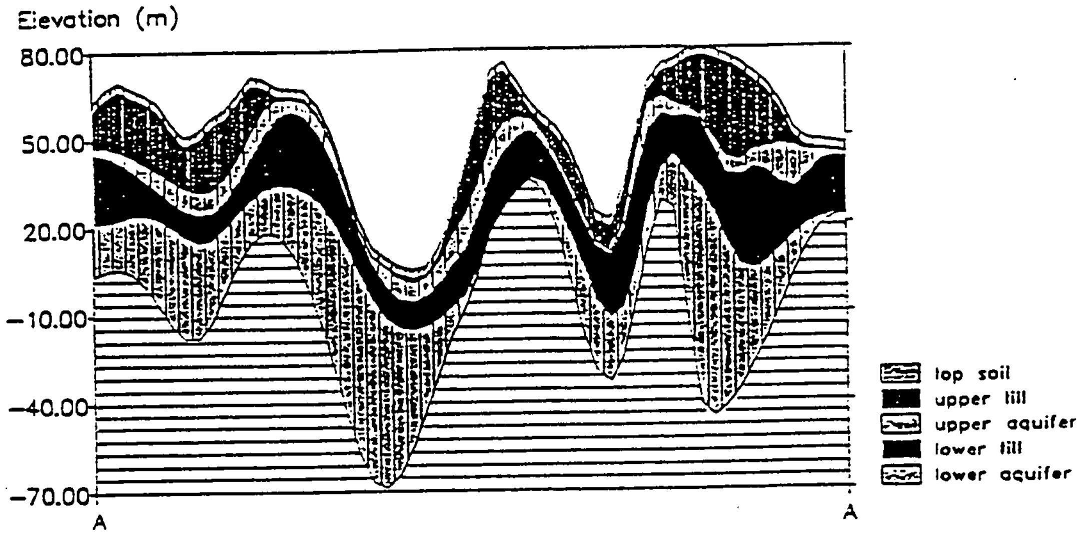
Fig. 2



level (m)
ABOVE 100.0
90.0 - 100.0
80.0 - 90.0
70.0 - 80.0
60.0 - 70.0
50.0 - 60.0
40.0 - 50.0
30.0 - 40.0
20.0 - 30.0
10.0 - 20.0
0.0 - 10.0
BELOW 0.0
A
A
10 20 30 40 50 60 70 80 90 100 110 120 130 140 150 160
10 20 30 40 50 60 70 80 90 100 110 120 130 140

Fig. 3


Elevation (m)
80.00
50.00
20.00
-10.00
-40.00
-70.00
A
A
top soil
upper till
upper aquifer
lower till
lower aquifer

82%

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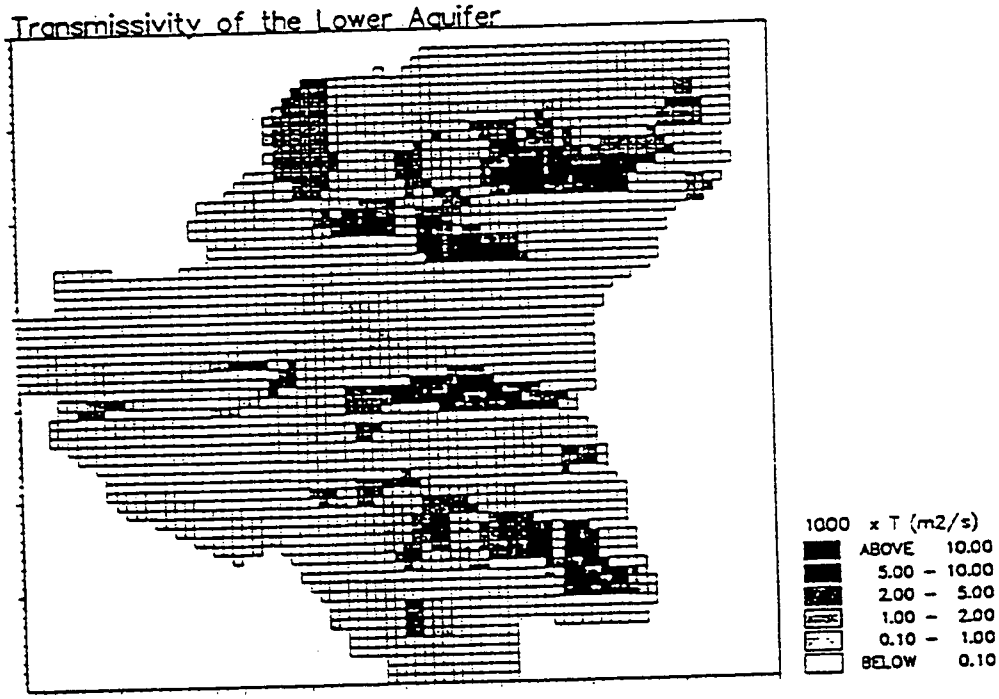
Fig. 4



Transmissivity of the Lower Aquifer
1000 x T (m2/s)
ABOVE 10.00
5.00 - 10.00
2.00 - 5.00
1.00 - 2.00
0.10 - 1.00
BELOW 0.10

Fig. 5

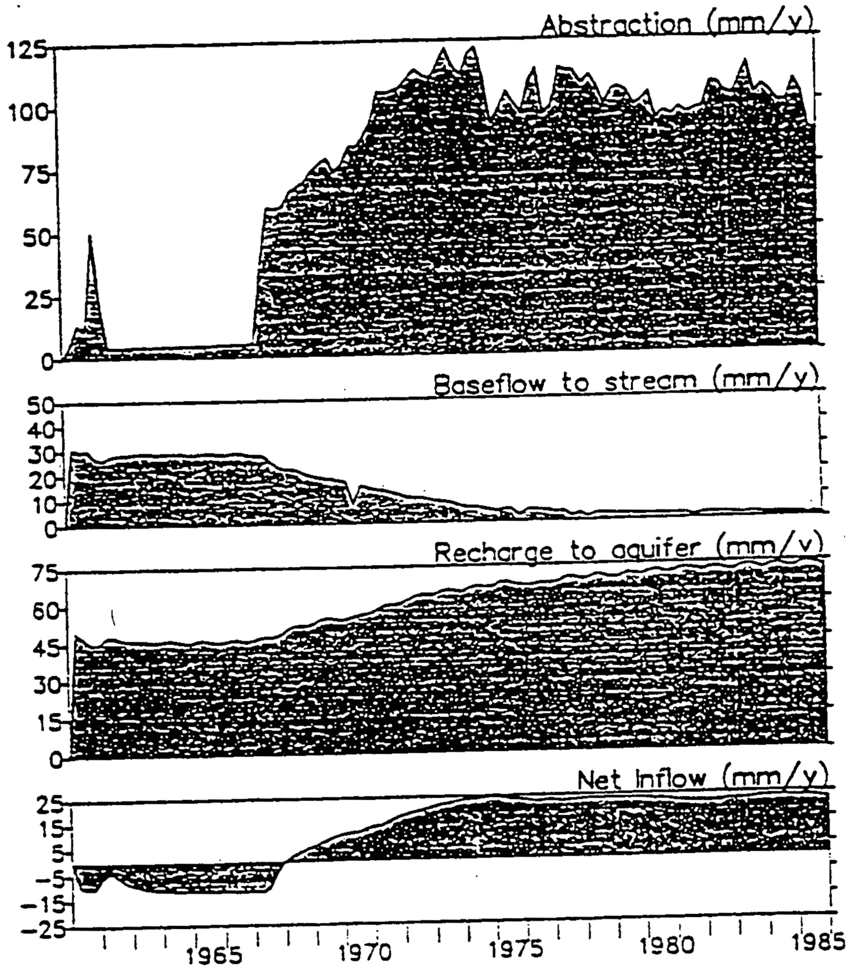


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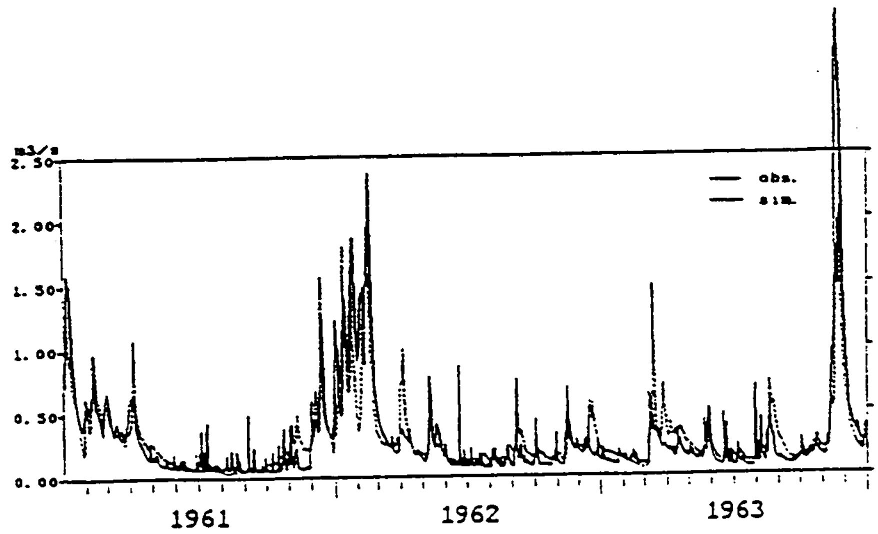
Fig. 6



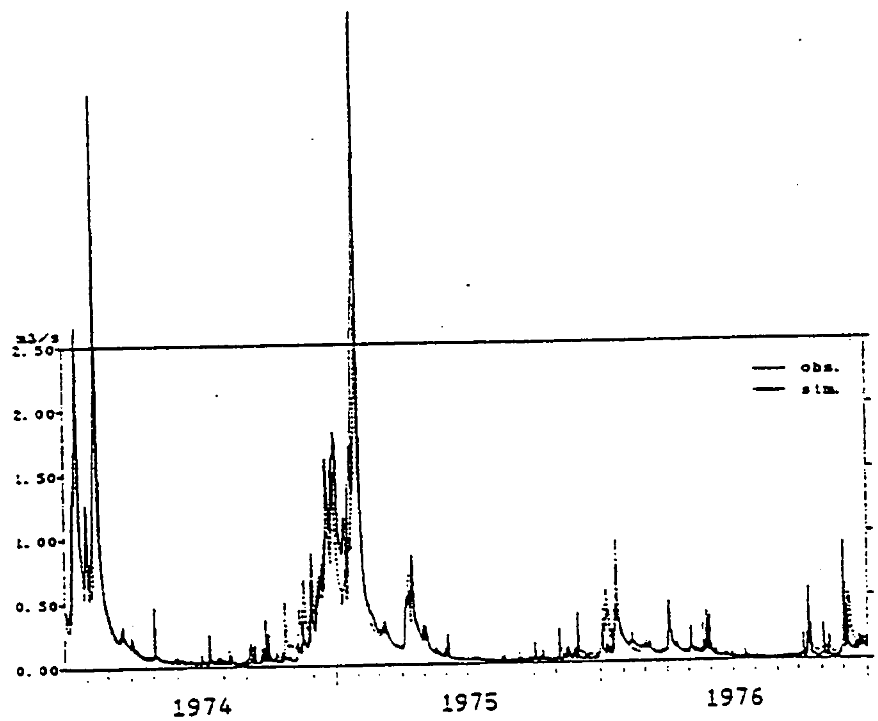


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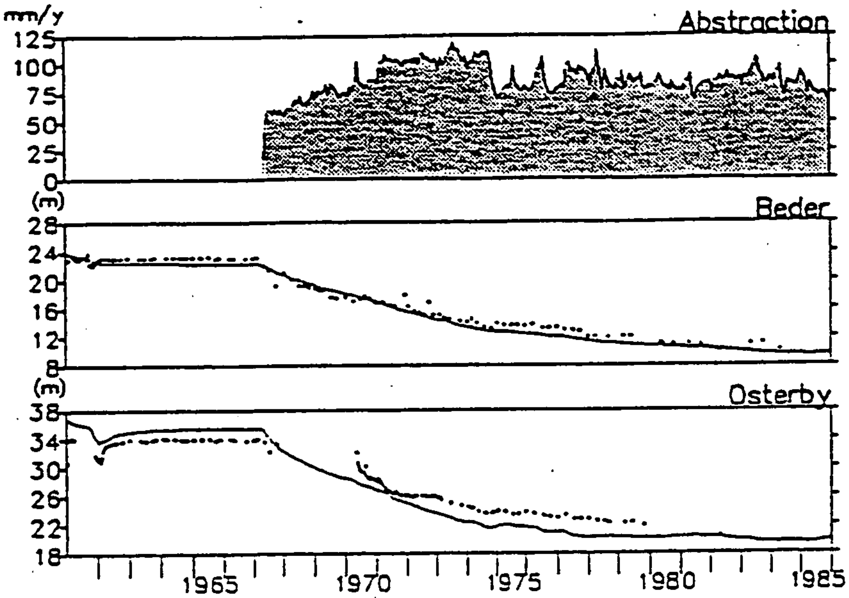
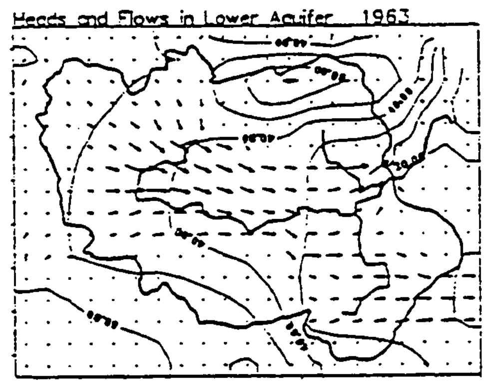
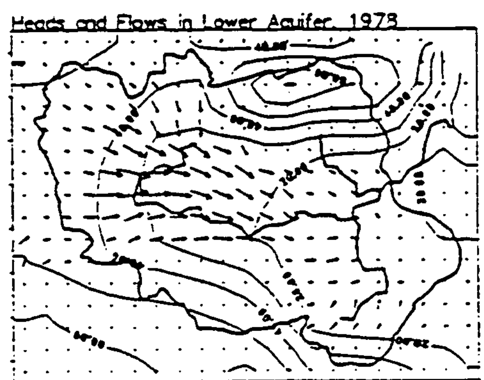
Fig. 7



mm/y
Abstraction
125
100
75
50
25
0
(m)
Beder
28
24
20
16
12
8
(m)
Osterby
38
34
30
26
22
18
1965
1970
1975
1980
1985



Heads and Flows in Lower Aquifer 1963



Heads and Flows in Lower Aquifer 1978

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Fig. 8

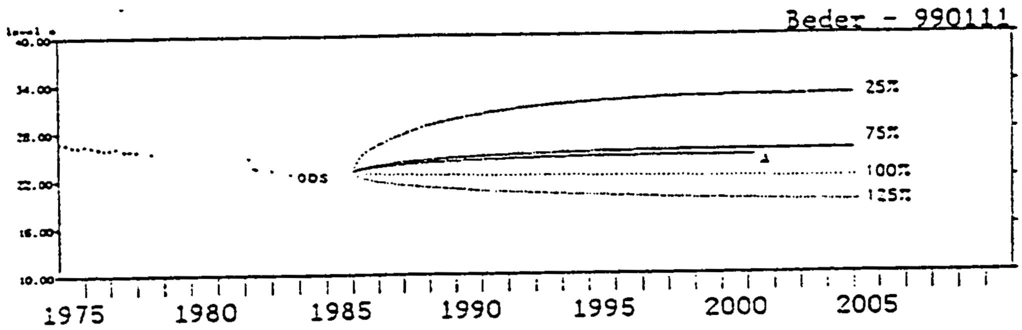

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Fig. 9

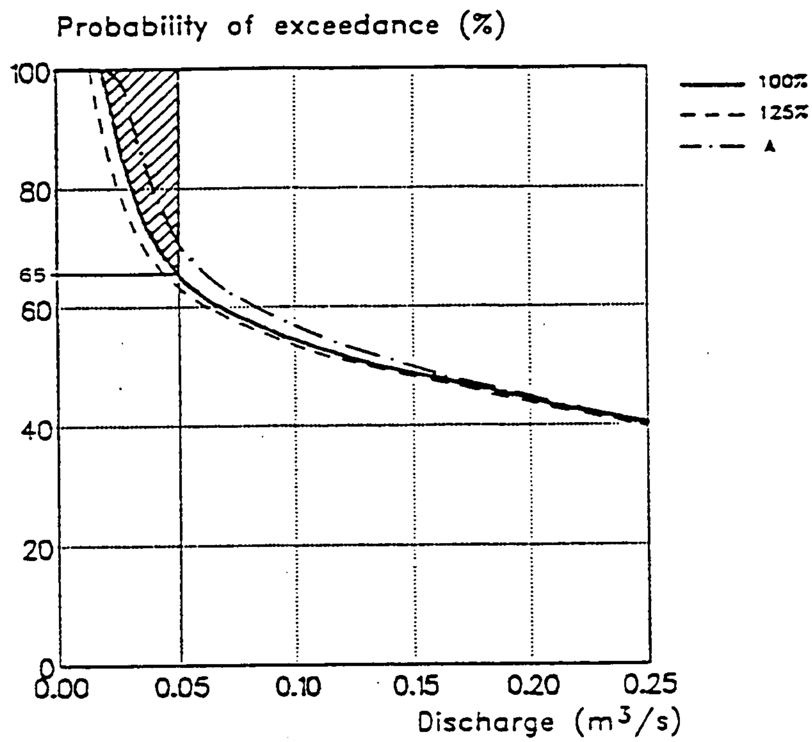
Probability of exceedance (%)
100
80
65
60
40
20
0
0.00
0.05
0.10
0.15
0.20
0.25
Discharge (m^3/s)
100%
125%
A

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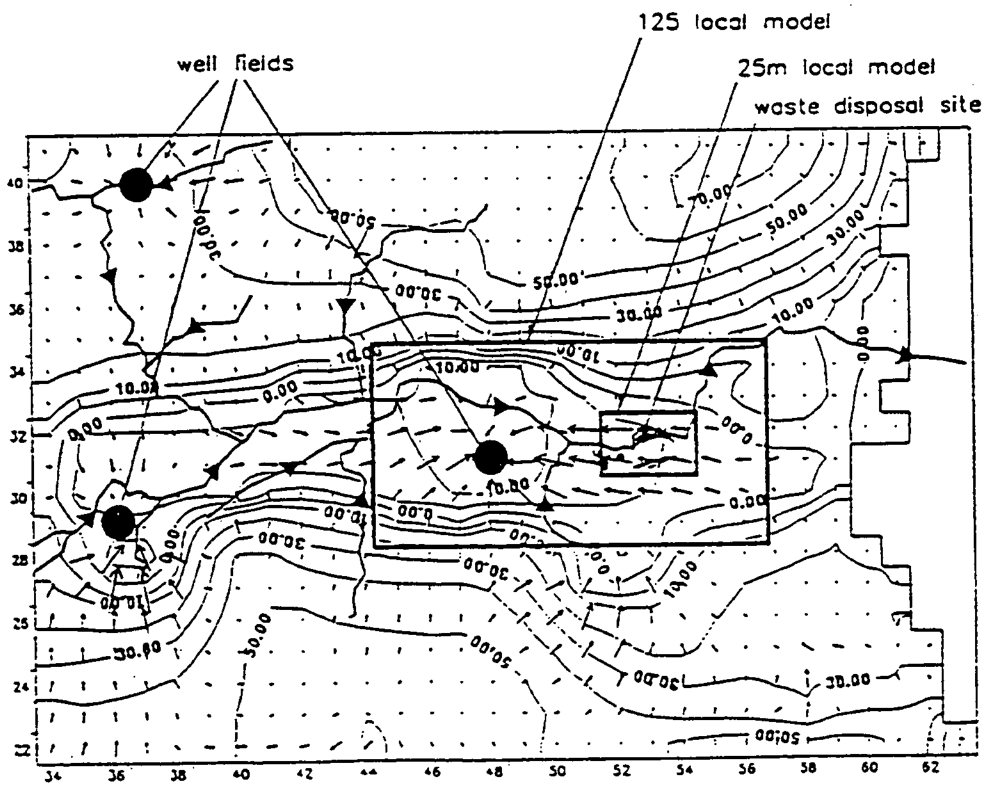
Fig. 10



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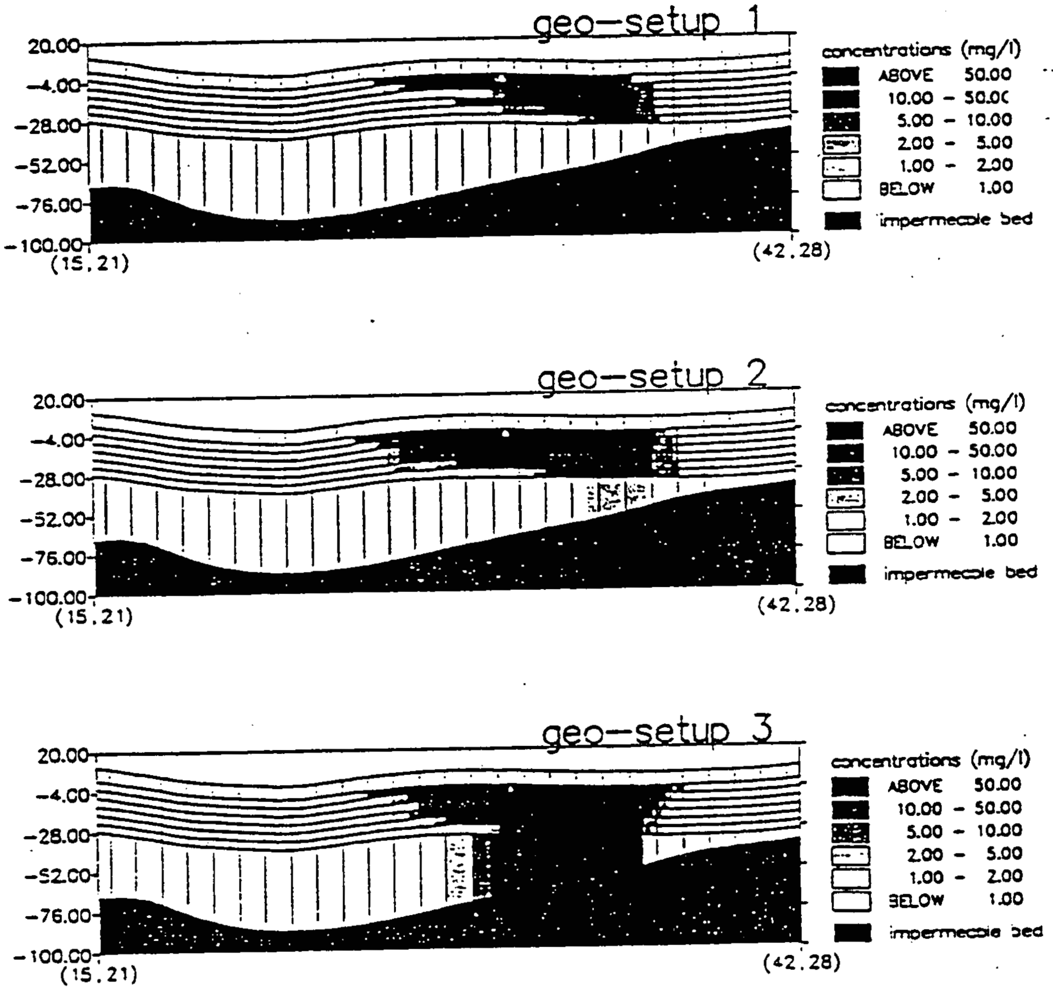
Best Available Copy

Fig. 11


geo-setup 1
20.00
-4.00
-28.00
-52.00
-76.00
-100.00
(15,21)
(42,28)
concentrations (mg/l)
ABOVE 50.00
10.00 - 50.00
5.00 - 10.00
2.00 - 5.00
1.00 - 2.00
BELOW 1.00
impermeable bed
geo-setup 2
geo-setup 3

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VANDINDVINDINGENS INDFLYDELSE PÅ GRUNDVANDSDANNELSE OG VANDLØBSAFSTRØMNING
Et eksempel fra Odense

Civilingeniør, Ph.D. Anders Refsgaard
Dansk Hydraulisk Institut

ATV Møde

Grundvandsdannelse - kvantitet og kvalitet

Danmarks Tekniske Højskole
Tirsdag den 4. maj, 1993

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INDLEDNING OG PROBLEMSTILLING

Debatten omkring vandindvinding i byområder, hvor der generelt er større risiko for forurening af grundvandsmagasiner, idet byområder ofte fungerer som en fladebelastning i forureningsmæssig henseende, har efterhånden stået på i nogen tid. Tilskyndet af dette samt konkrete forureningsopdagelser og den generelle forsyningssikkerhed for vandforsyningen til Odense Kommunes borgere, besluttede Fyns Amt og Odense Vandforsyning at iværksætte en undersøgelse af konsekvenserne af at anlægge en ny kildeplads ca. 15 km syd for Odense by. Kildepladsen skal dimensioneres til en indvinding på ca. 4 mill. m^3/år svarende til 25% af den nuværende indvinding fra Odense Kommunale Værkers kildepladser.

En indvinding i denne størrelsesorden vil naturligvis påvirke vandbevægelserne i det hydrologiske kredsløb markant indenfor et vist område fra kildepladsen. Fyns Amt har fokuseret på problemstillingerne omkring påvirkninger dels af forholdene i særlige beskyttelsesområder - primært bestående af vådområder og hermed vandløb og vandløbsomgivelser - dels af potentialeforholdene i de primære og sekundære grundvandsmagasiner.

Problemstillingen kunne konkretiseres som en undersøgelse af effekterne på:

* minimumsvandføringerne i berørte vandløb
* forholdene i berørte vådområder
* potentialeforholdene i primære og sekundære grundvandsmagasiner

ved alternative placeringer af den nye kildeplads samt alternative indvindingsstrategier fra nogle af de eksisterende kildepladser.

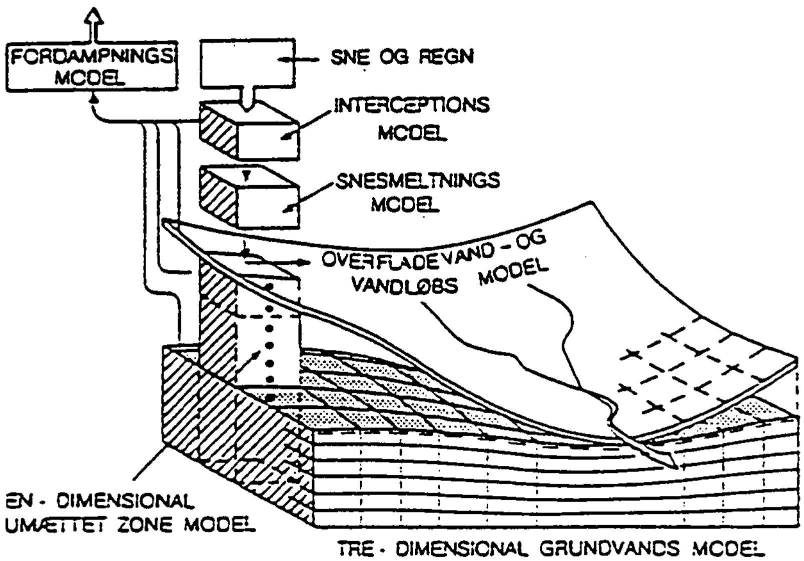
Af Suså-undersøgelsen (Refsgaard og Hansen, 1982), hvor problemstillingen omkring reduktion af vandføringer på grund af grundvandsindvinding også blev behandlet, fremgår det, at selv i områder med tilsyneladende ensartede hydrologiske og hydrogeologiske forhold er der stor stedlig variation i påvirkningsgraden. Af samme grund konkluderedes det, at "simple" vandbalanceberegninger ikke fører til realistiske vurderinger af påvirkningsgraden. Der er derfor i nærværende undersøgelse anvendt en integreret hydrologisk model til at kvantificere forholdene mellem grundvandsdannelse og vandløbsafstrømning.

KVANTIFICERING AF PROBLEMSTILLINGEN

Beskrivelse af modelopstillingen

Som grundlag for kvantificeringen af ovenstående problemstilling er den hydrologisk model **SHE** anvendt, idet den kan beskrive de vigtigste strømningsprocesser i landfasen i det hydrologiske kredsløb; nedbør/fordampning, strømning i den umættede zone og i flere grundvandsmagasiner samt strømning i overfladerecipienter og vandudveksling mellem de forskellige dele af kredsløbet. Modellen kan skematisk fremstilles som vist i Figur 1.

SHE-modellen er en såkaldt distribueret model, som i hovedtræk er baseret på en fysisk korrekt matematisk beskrivelse af de relevante processer. I praksis vil det sige, at alle parametre som for eksempel topografiske forhold og hydrauliske ledningsevner kan specificeres som værdier fordelt over beregningsområdet, og at disse værdier er fysisk målelige størrelser. Resultaterne fås ligeledes som værdier (potentialer, vandføringer etc.) fordelt over beregningsområdet og over modellens simuleringsperiode.



Figur 1 Skematisk fremstilling af SHE-modellen

Modellen er opstillet af to omgange dels som en regional model, som dækker hele det topografiske opland til Odense Å, hvilket svarer til 925 km² eller ca 1/3 af Fyn, dels som en lokal model, som dækker de mest interessante områder syd og sydvest for Odense by, og hvis omtrentlige udstrækning fremgår af Figur 2. Den regionale model danner i denne sammenhæng basis for beskrivelsen af randbetingelserne for den lokale model i form af potentialer og vandføringer.

Datagrundlaget for opstillingen af modellen har omfattet 20 års meteorologiske data i form af nedbør, fordampning og temperatur, topografiske data, jordfysiske og geologiske data i form af jordprofiler, retentionskurver, geologiske profiler og transmissivitetsforhold, tidsserier for vandindvinding, vegetationsparametre etc. samt geometriske data for vandløbene i form af tværsnit og bred- og bundkoter.

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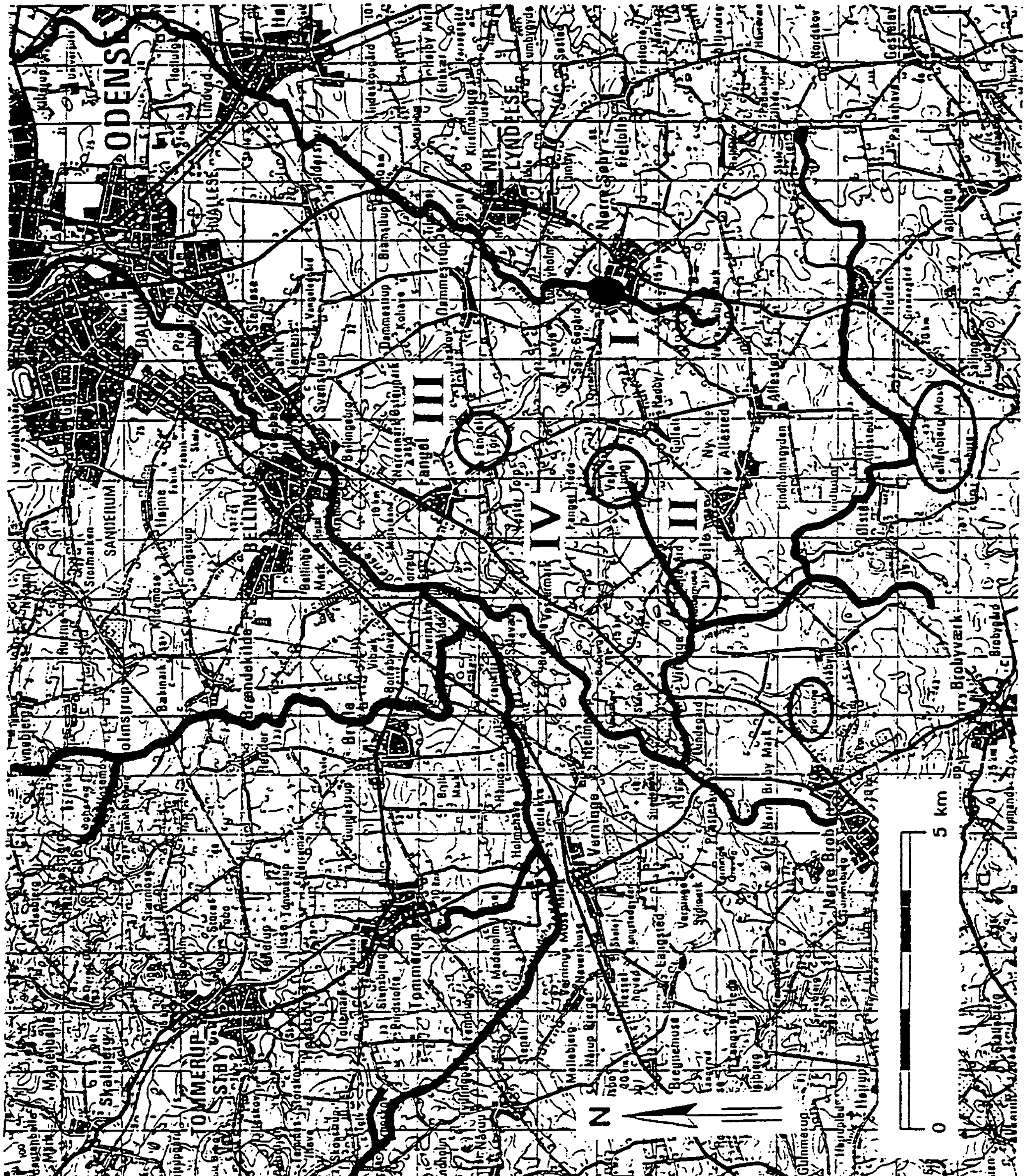

Figur 2 Modelområde med indtegning af nøgleområderne. De anførte romertal referer til alternative placeringer af den nye kildeplads, mens de særlige beskyttelsesområder ud over vandløbene er markeret med en ring

De kvartære geologiske aflejringer er præget af regionale smeltevandsaflejringer, der som regel kan udnyttes som højtydende grundvandsmagasiner. Der er som sådan ikke de store problemer med at finde egnede områder til grundvandsindvinding ud fra en forsyningsmæssig indgangsvinkel.

I den matematiske model er geologien simplificeret til en 5-lags model med to grundvandsmagasiner adskilt og overlejret af aquitarder og med et tyndt lag øverst, som indeholder den umættede zone.

Den hydrologiske model er kalibreret og valideret mod tidsserier af vandløbsafstrømning og grundvandspotentialer for perioden 1972 til 1992 - en periode, som indeholder forskellige (ekstreme) hydrologiske hændelser som for eksempel en periode med meget tørre somre (1975/76) og den meget våde periode 1981/82. Den samme periode er anvendt i konsekvensvurderingerne, i det kun de påtrykte påvirkninger i form af vandindvindinger er antaget konstante i tiden (dog således, at variationen i indvindingen fra de enkelte kildepladser over året bibeholdes).

Konsekvenser af alternative placeringer af den nye kildeplads

Der er tidligere blevet udført både geofysiske kortlægninger, prøveboringer og -pumpninger i det område, som synes mest attraktivt for placeringen af den nye kildeplads, som er benævnt I i Figur 2. Disse undersøgelser har ført frem til en foreløbig mest gunstig placering primært ud fra grundvandshydrauliske og forsyningsmæssige hensyn. Samtidig er der peget på tre alternative placeringer af den nye kildeplads dels ud fra forureningsmæssige betragtninger (placeringen af eksisterende og kommende forureningstruende lokaliteter) dels ud fra indledende betragtninger om påvirkningerne af de udpegede særlige beskyttelsesområder. Placeringen af de 4 mulige kildepladser (I - IV) samt de særlige beskyttelsesområder fremgår af Figur 2.

Ved gennemregning af de forskellige kildepladsplaceringer med en indvinding på 4 mill. m^3/år blev det hurtigt klart, at kun placering I og III ville være alternativer, idet der simpelthen ikke var "vand nok" i magasinet ved placering II og IV. I modelmæssig sammenhæng giver dette sig udslag i, at den beregningskasse, hvori indvindingen er påtrykt "løber tør" for vand.

De **overordnede** vandbalancer for det hydrologiske kredsløb er i praksis identiske for kildepladsplacering I og III. Figur 3 viser således vandbalancerne i referencesituationen uden indvinding og ved de to kildepladsplaceringer I og III. Det fremgår af figuren, at den forøgede indvinding, som svarer til 14 mm, modsvares af en forøget nedsivning til det primære magasin, en reduceret nettoudstrømning over modelranden og en formindsket opstrømning fra det primære magasin til vandløbene ("baseflow"). Den forøgede nedsivning fra de øvre magasiner giver sig udslag i en reduktion i tilstrømningen til vandløbene fra disse magasiner og i en lidt større nettonedbør og dermed mindre fordampning.

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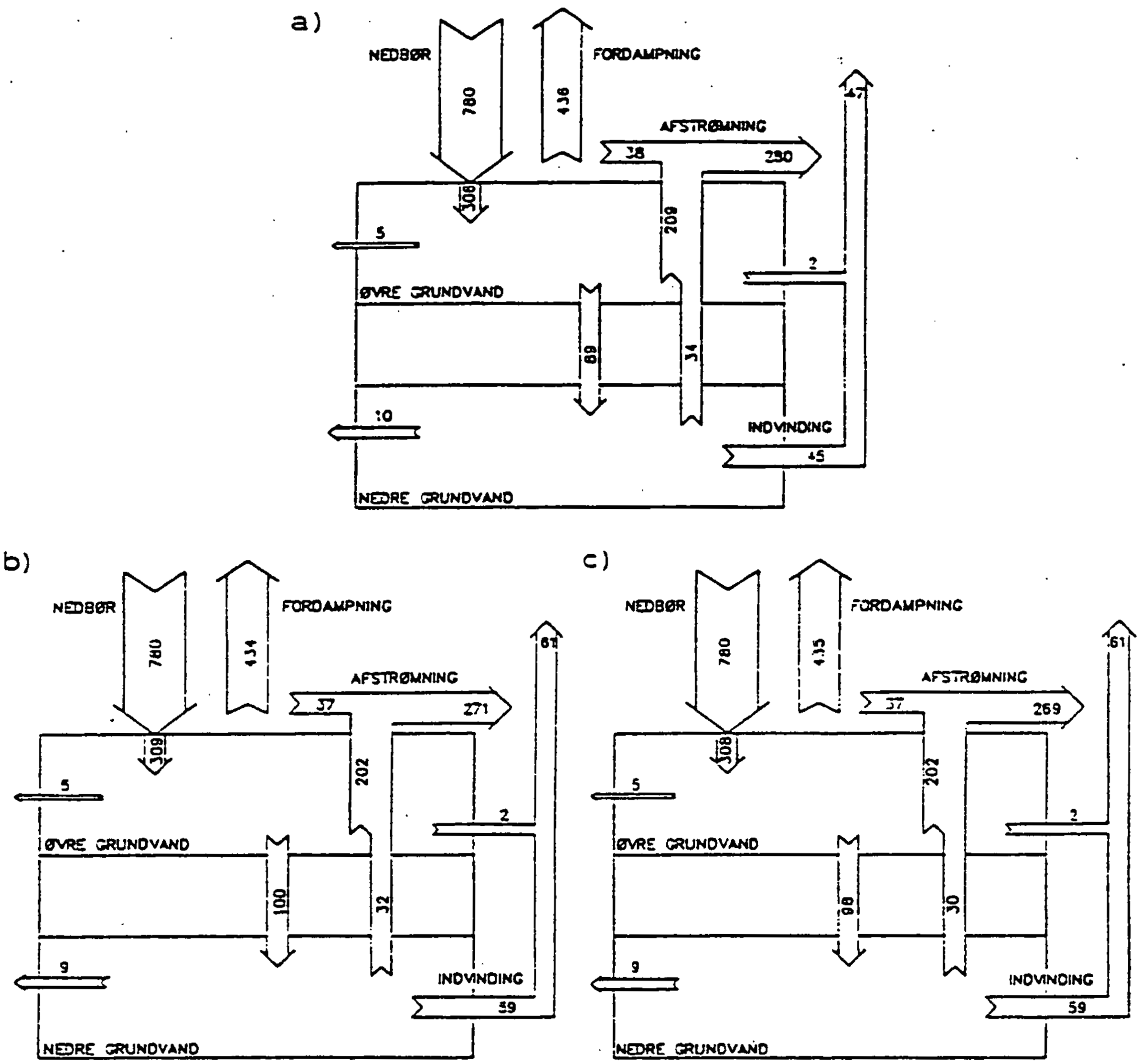


Figur 3 Vandbalancer beregnet som middel over hele perioden (alle tal i mm/år)
a: referencesituation b: kildepladsplacering I c: kildepladsplacering III

Ved en nærmere analyse af de beregnede afstrømningshydrografer for nærliggende vandløb viser det sig, at de små forskelle i baseflow ved de to kildepladsplaceringer (32 og 30 mm/år) dækker over store forskelle i påvirkningsgraden, P, som kan defineres som:

$$P = \frac{\Delta Q_{min.}}{\Delta Q_{mid.}}$$

hvor P er påvirkningsgraden
$\Delta Q_{min.}$ er ændringen i minimumsvandføringen
$\Delta Q_{mid.}$ er ændringen i middelafstrømningen.

P er beregnet for de 3 vandløb, som omkranser den nye kildeplads, og anført i Tabel 1. Indindingen på 4 mill. m³/år svarer til 126 l/s.

	Odense Å station 15 km		Lindved Å station 1 km		Vittinge Å station 0 km	
Kildepladsplacering	P_{min} [-]	ΔQ_{mid} [l/s]	P_{min} [-]	ΔQ_{mid} [l/s]	P_{min} [-]	ΔQ_{mid} [l/s]
I	0.2-0.4	58	0.03-0.08	39	0.0-0.08	24
III	0.55-0.68	81	0.0-0.10	10	0.0-0.20	5

Tabel 1 Vandløbspåvirkninger fra de forskellige kildepladsplaceringer

Det fremgår af tabellen, at der er store variationer både i tid og sted af påvirkningsgraden. Der er dog temmelig store usikkerheder ved beregningen af påvirkningsgraden i Lindved Å og Vittinge Å, idet ændringer i minimumsvandføringerne er så små at de nærmer sig usikkerheden på beregningen. Selv om de to kildepladser ligger med en indbyrdes afstand på mindre end tre kilometer fra hinanden er der stor forskel i påvirkningen fra de to kildepladsplaceringer.

Kildepladserne er placeret i højtransmissive aflejringer, hvilket bevirker, at grundvandssænkninger spredes ud over et stort område. Hermed bliver påvirkningen af de særlige beskyttelsesområder begrænset, idet der opretholdes opadrettede gradienter til vådområderne under de fleste forhold.

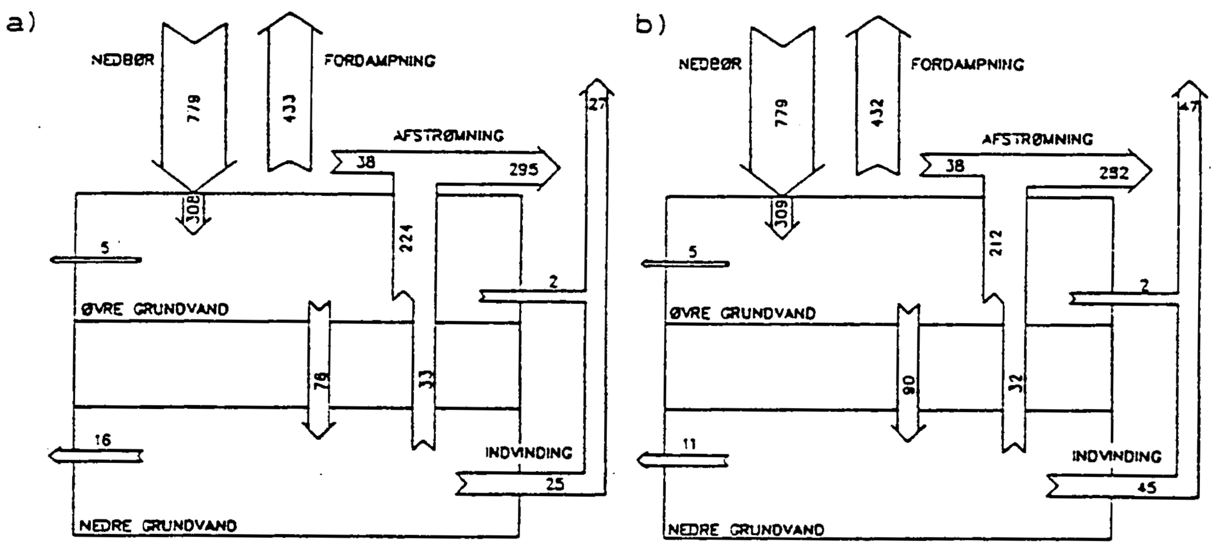
Konsekvenser af alternative indvindingsstrategier

På Odense Vandforsynings indvindingsområder ved Holmehave, som ligger ved Holmehave Bæk, og Borreby, som ligger ved Borreby Møllebæk indvindes der i øjeblikket ca. 4.6 henholdsvis ca. 1.3 mill. m³/år. Med etableringen af den nye kildeplads vil det eventuelt komme på tale, at drosle ned på indvindingen fra den ene eller begge kildepladser, hvis det giver en væsentlig forbedring af afstrømningsforholdene i de nærliggende vandløb. På denne baggrund er der udvalgt de i Tabel 2 viste 5 scenarier, idet referencekørslen udgøres af et beregning uden indvinding fra de to områder i hele perioden. Der er i beregningerne ikke regnet med en indvinding fra det nye indvindingsområde omtalt ovenfor.

Indvindingssituation	Indvinding fra Holmehave (mill. m³/år)	Indvinding fra Borreby (mill. m³/år)
Referencesituation	0.0	0.0
A	4.6	1.3
B	0.0	1.3
C	4.6	0.0
D	2.6	1.3
E	1.0	1.3

Tabel 2 Oversigt over gennemregnede scenarier

De overordnede, gennemsnitlige vandbalancer er vist i Figur 4 for referencesituationen og indvindingssituation A. Som det ses af figuren er vandbalancen for de to situationer næsten identisk for de "overjordiske" komponenters vedkommende, idet mere end halvdelen af de 779 mm som udgør nedbøren fordamper, mens kun godt 300 mm infiltrerer til grundvandet og knap 40 mm løber direkte til vandløbene.



Figur 4 Vandbalancer
a: referencesituation b: indvindingssituation A

I referencesituationen udgør infiltrationen til de primære grundvandsmagasiner 76 mm, hvoraf 25 mm udnyttes til vandindvinding, 16 mm udgør nettoudstrømningen over modelafgrænsningerne og 33 mm udgør "baseflowet" til vandløbene. Af de godt 300 mm infiltrerende vand strømmer ca. 2/3 tilbage til vandløbene gennem drænsystemer og de øvre grundvandsmagasiner.

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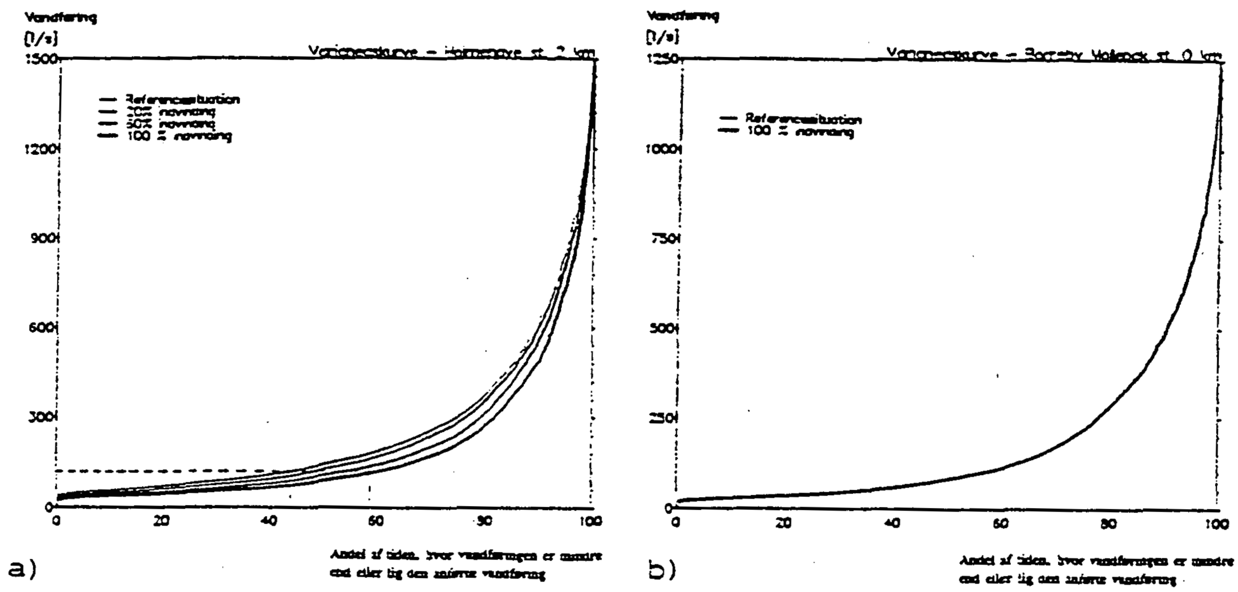
En indvinding på i alt 5.9 mill. m^3/år fra områderne ved Holmehave og Borreby svarende til 20 mm giver anledning til en noget større infiltration til de primære grundvandsmagasiner og dermed en mindre tilstrømning til vandløbene fra de øvre grundvandsmagasiner. Derimod er "baseflowet" fra de nedre grundvandsmagasiner næsten uændret.

En nærmere undersøgelse viser dog, at påvirkningen af vandløbene er meget forskellige. Figur 5 viser således påvirkningen af Holmehave Bæk og Borreby Møllebæk for de forskellige scenarier i form af varighedskurver for vandføringen. Figuren viser, at afstrømningen fra Borreby Møllebæk er næsten upåvirket af tilstedeværelsen af vandværket, mens Holmehave Bæk påvirkes mærkbart. Minimumsvandføringerne påvirkes i nogen grad, men det er især de mellemstore vandføringer (100 - 400 l/s), som påvirkes af de forskellige indvindinger fra kildepladsen. F.eks. vil en vandføringen i bækken være mindre end 120 l/s 43% af tiden, hvis der ingen indvinding foregår på Holmehave kildeplads, mens det tilsvarende tal er 57%, hvis den nuværende indvinding fastholdes.



Figur 5 Varighedskurver for vandføringen
a: Holmehave Bæk b: Borreby Møllebæk

Påvirkningerne af minimumsvandføringen i Holmehave Bæk kan også kvantificeres ved påvirkningsgraden, P, som er beregnet og anført i Tabel 3.

Tabellen viser, at der er stor forskel i påvirkningsgraden på de 2 vandløb. Det skal dog bemærkes, at den beregnede påvirkningsgrad i Borreby Møllebæk er behæftet med temmelig stor usikkerhed på grund af de små vandføringsændringer, der er tale om. Samtidig har beregningerne vist, at der er forholdsvis stor variation af påvirkningsgraden over de forskellige år.

	Holmehave Bæk station 2.0 km			Borreby Møllebæk station 0 km		
Indvindingssituation	$\Delta Q_{indv.}$ [l/s]	$P_{min.}$ [-]	$\Delta Q_{mid.}$ [l/s]	$\Delta Q_{indv.}$ [l/s]	$P_{min.}$ [-]	$\Delta Q_{mid.}$ [l/s]
A	0	0.25-0.75	4	41	0.14-0.28	7
B	146	0.22-0.34	62	0	*	< 1
C	83	0.26-0.40	42	41	0.14-0.28	7
D	32	0.28-0.61	18	41	0.14-0.28	7

* Vandføringsændringerne er så små, at P ikke kan beregnes realistisk.

Tabel 3 Vandløbspåvirkningen fra de forskellige indvindingssituationer

Tabellen viser endvidere, at påvirkningsgraden er afhængig indvindingen, idet en lille indvinding giver sig udslag i en større påvirkningsgrad, hvilket igen hænger sammen med, hvor stor en del af den indvundne vandmængde, der stjæles fra naboområderne.

KONKLUSIONER

Det hydrologiske kredsløb i et område syd for Odense er blevet undersøgt ved at konceptualisere de indgående processer i et matematisk prognoseværktøj i form af den hydrologiske model SHE. Modellen er efterfølgende blevet anvendt til at vurdere konsekvenserne af forskellige indgreb i kredsløbet. Disse har dels været at etablere en ny kildeplads med en kapacitet på 4 mill. m^3/år, dels at neddrosle af indvindingen på kildepladserne ved Holmehave og Borreby.

Modelundersøgelserne har givet anledning til følgende hovedkonklusioner:

* Indvinding fra højtydende grundvandsmagasiner med regional udstrækning vil i nogle tilfælde betyde, at grundvandssænkningerne bliver fordelt så meget, at påvirkningerne på både nærliggende vandløb og vådområder vil blive akceptable.

* Der er en stor tidslig og stedlig variation i påvirkningsgraden af forskellige vandløb fra grundvandsindvinding. En stor grundvandsindvinding fra nedre grundvandsmagasiner behøver ikke nødvendigvis at betyde en tilsvarende stor reduktion i medianminimumsvandføringen, idet den forøgede infiltration til de nedre magasiner kan "stjæles" fra de mellemstore vandføringer, som primært stammer fra de terrænnære magasiner.

* Generelt er der ikke en lineær sammenhæng mellem indvinding og påvirkningsgraden.

* Et prognoseværktøj er efter kalibreringen nemt at anvende til at vurdere forskellige indgreb i det hydrologiske kredsløb og må i det hele taget betragtes som en nødvendighed til en sådan vurdering, idet variationen i tid og sted af de forskellige styrende faktorer ikke lader sig vurdere uden avancerede matematiske modeller.

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MODELLING THE INFLUENCES OF GABCIKOVO HYDRO POWER PLANT ON THE HYDROLOGY AND ECOLOGY OF THE DANUBIAN LOWLAND

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Introduction

The Danubian Lowland between Bratislava and Komárno is an inland delta formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which throughout the year receives in the order of 25 m^3/s infiltration water from the Danube in the upper parts of the area and returns it into the Danube and the drainage channels in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

Human influence has gradually changed the hydrological regime in the area. Construction of dams upstream of Bratislava together with exploitation of river sediments has significantly deepened the river bed and lowered the water level in the river. These changes have had a significant influence on the conditions of the ground water regime as well as the sensitive riverside forests downstream of Bratislava. In spite of this basically negative trend the floodplain area with its alluvial forests and the associated ecosystems still represents a very unique landscape of outstanding importance.

The construction of the hydraulic structures in connection with the hydropower plant at Gabcikovo also significantly affects the hydrological regime and the ecosystem of the region.

To utilize state-of-the-art modelling technology for addressing the water resources problems in the area the project "Danubian Lowland - Ground Water Model" has been defined within the PHARE programme agreed upon between the Commission of the European Communities and the Government of the Slovak Republic.

A brief description of the on-going PHARE project "Danubian Lowland - Ground Water Model" is presented. The overall objective is to provide a comprehensive technical/scientific based management tool for the water resources decision makers. An integrated mathematical modelling system describing flows, water quality processes, sediment transport/erosion in river, flood plain, reservoir and ground water system is being developed. This system will be fully coupled with a large data base and GIS system.

A key for analysing the effects on the water resources of the changed surface water conditions caused by the Gabcikovo plant on the subsurface system is an integrated description of the river-aquifer system. The paper provides details on the modelling of the river branch system and the aquifer system, which is being carried out by developing a full coupling between DHI's two modelling systems for rivers and hydrology, MIKE 11 and MIKE SHE respectively.

The paper presents some preliminary model results. Further results will be presented at the conference.

1. Objective and framework of the PHARE project

To understand and analyze the complex relationships between physical, chemical and biological changes in the surface- and subsurface water regimes requires multidisciplinary expertise in combination with advanced mathematical modelling techniques.

The overall project objective is to establish a reliable impact assessment model for the Danubian Lowland area, which enables the authorities to formulate optimal management strategies leading to a protection of the

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water resource and a sound ecological development for the area.

The PHARE project is being executed by the Slovak Ministry of the Environment. Specialists from the following Slovakian organisations are involved in various aspects of the project implementation:

* Comenius University, Faculty of Natural Science (PRIF UK)
* Water Research Institute (VUVH)
* Irrigation Research Institute (VUZH)

A Danish-Dutch consortium of six organisations was selected as consultant for the project.

The project was initiated in the beginning of 1992 and will be completed by the end of 1995. At present the necessary equipment has been delivered to the project in Bratislava and the modelling and information system has been installed on computer workstations. The modelling work has been initiated; however no final model calibrations nor predictions have been carried out at this stage.

2. Establishment of Danubian Lowland Information System (DLIS)

An automated system is presently being developed to support the modelling activities. The integrated modelling system will be interfaced to a central information system. The central information system, called Danubian Lowland Information System (DLIS), will provide the different models with the necessary data and functionality to elaborate further on the modelling results.

Because of the complexity and the amount of data involved - first estimates indicate about 2 Gbyte of data - major attention has been paid to the development of the underlying data model. The information needed for the modelling originates from different monitoring networks, which are maintained and observed by different institutes. The larger part of the data is available from automated archives and can be loaded into the system from magnetic medium.

The information to be incorporated in DLIS has a pronounced spatial character. The two main components of the GIS are a geographical information system (GIS), for which ARC/INFO has been selected, and a relational data base management system (RDBMS), for which INFORMIX has been selected.

3. Establishment of integrated modelling system

The integrated modelling tool, which will form the basis for all the modelling activities, is based on the following packages which can be used individually or brought together in an integrated manner:

* MIKE SHE which, on catchment scale, can simulate the major flow and transport processes of the hydrological cycle which are traditionally divided in separate components:
  - 1-D flow and transport in the unsaturated zone
  - 3-D flow and transport in the ground water zone
  - 2-D flow and transport on the ground surface
  - 1-D flow and transport in the river.

  All the above processes are fully coupled allowing for feedbacks and interactions between components. In addition to the above mentioned components, MIKE SHE includes modules for multi-component chemical reactions in the unsaturated and saturated zone as well as a component for oxygen consumption and transport in the unsaturated zone.
* MIKE 11, which is a one-dimensional river modelling system. MIKE 11 is used for hydraulics, sediment transport and morphology, and water quality. The modules for sediment transport and morphology are able to deal with cohesive and non-cohesive sediment transport, as well as the accompanying morphological changes of the river bed. The non-cohesive model will operate on a number of different grain sizes, taking into account shielding effects. The cohesive model deals both with consolidation of the river bed and flocculation.
* MIKE 21, which is a two-dimensional hydrodynamic modelling system. MIKE 21 is used for reservoir modelling, including hydrodynamics, sediment transport and water quality. The sediment transport modules deals with both cohesive and non-cohesive sediment, and the non-cohesive module will operate on a number of different grain size fractions.
* Both of the above mentioned models include River/Reservoir Water Quality (WQ) and

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Eutrophication (EU) modules to describe oxygen, ammonium, nitrate and phosphorus concentrations and oxygen demands as well as eutrophication issues.

* DAISY is a one-dimensional root zone model for simulation of crop production, soil water dynamics, and nitrogen dynamics in crop production for various agricultural management practices and strategies. The particular processes considered include transformation and transport involving water, heat, carbon and nitrogen.

The above mentioned models are all generalized tools with comprehensive applicability ranges, and they are well proven in a large number of international projects. In addition, some model modifications are being carried out during the project in order to accommodate the very special environment and problems observed in the area.

With regard to simulation of floodplain hydrology and ecology the core of the integrated modelling system is constituted by the MIKE SHE, the MIKE 11 and a newly developed, full coupling of the two systems as described in the following three sections.

3.1 MIKE SHE

The European Hydrological System - SHE was developed in a joint effort by Institute of Hydrology (UK), SOGREAH (France) and Danish Hydraulic Institute. It is a deterministic, fully-distributed and physically-based modelling system for describing the major flow processes of the entire land phase of the hydrological cycle. A description of the SHE is given in [Abbott et al, 1986[1] and 1986[2]]. Since 1987 the SHE has been further developed independently by the three respective organizations which now are University of Newcastle (UK), Laboratoire d'Hydraulique de France and DHI. DHI's version of the SHE, known as the MIKE SHE, represents significant new developments with respect to user interface, computational efficiency and process descriptions [DHI, 1993[3]].

MIKE SHE solves the partial differential equations for the processes of overland and channel flow, unsaturated and saturated subsurface flow. The model is completed by a description of the processes of snow melt, interception and evapotranspiration. The flow equations are solved numerically using finite difference methods.

In the horizontal plane the catchment is discretized in a network of grid squares. The river system is assumed to run along the boundaries of these. Within each square the soil profile is described in a number of nodes, which above the groundwater table may become partly saturated. Lateral subsurface flow is only considered in the saturated part of the profile. Fig. 1 illustrates the structure of the MIKE SHE. A description of the methodology and some experiences of model application are presented in [Refsgaard et al.1992[4]; Jain et al, 1992[5]; Lohani et al, 1993[6]; Styczen and Storm, 1993[7]].

The spatial and temporal variations in the catchment characteristics and meteorological input are provided in a series of two-dimensional matrices of grid square codes. To each code is further attached a number of attributes describing either parametric data or input data.

The distributed description in the MIKE SHE allows the user to include and test against spatially varying data. MIKE SHE is a multi output model which besides discharge in any river link also produces information about e.g. water table elevations, soil moisture contents, infiltration rates, evapotranspiration, etc. in each grid square.

3.2 MIKE 11

MIKE 11 is a comprehensive, one-dimensional modelling system for the simulation of flows, sediment transport and water quality in estuaries, rivers, irrigation systems and other water bodies. It is a 4th generation modelling package designed for micro-computers with DOS or UNIX operating systems and provides the user with an efficient interactive menu and graphical support system with logical and systematic layouts and sequencing in the menus. The package was introduced in 1989 and today the number of installations world-wide exceeds 300.

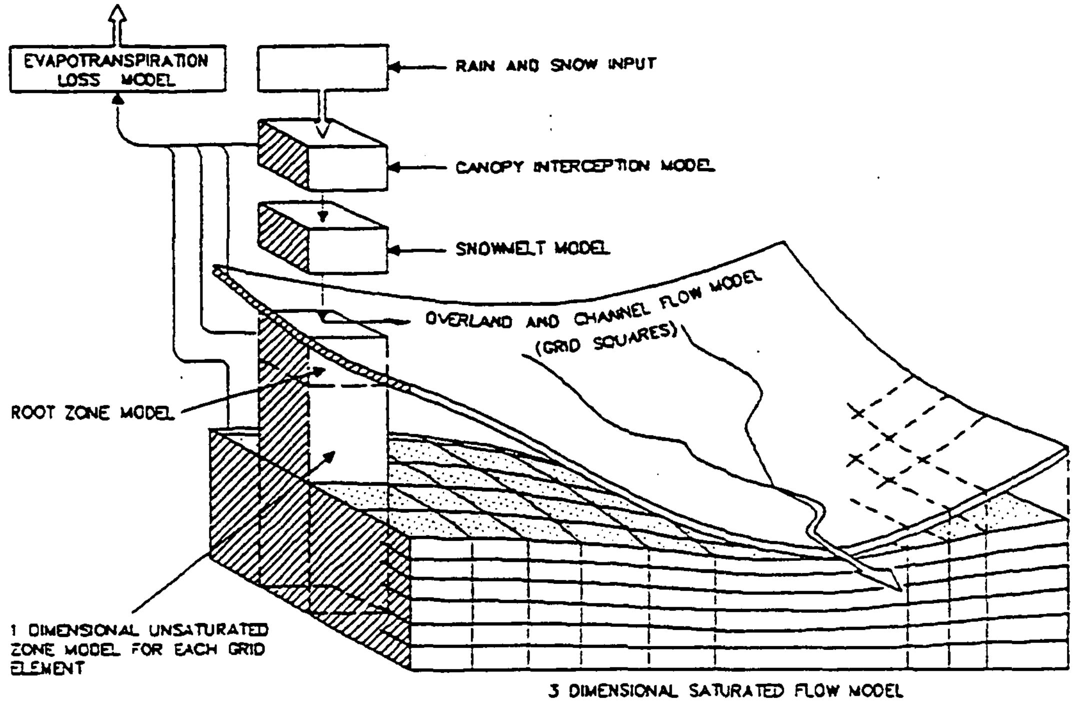

Fig. 1. Schematic presentation of the MIKE SHE.

The hydrodynamic module of MIKE 11 is based on the complete partial differential equations of open channel flow (Saint Venant). The equations are solved by implicit, finite difference techniques. The formulations can be applied to branched and looped networks and quasi two-dimensional flow simulations on floodplains.

MIKE 11 operates on the basis of information about the river and the floodplain topography, including man-made hydraulic structures such as embankments, weirs, gates, dredging schemes and flood retention basins. The hydrodynamic module forms the basis for morphological and water quality studies by means of add-on modules. A more detailed description of MIKE 11 is given in [DHI, 1993[a]]

3.3 A coupling of MIKE SHE and MIKE 11

The focus in MIKE SHE lies on catchment processes with a comparatively less advanced description of river processes. In contrary MIKE 11 has a more advanced description of river processes and a simpler catchment description than MIKE SHE. Hence, for cases where full emphasis is needed for both river and catchment processes a coupling of the two modelling systems is required.

A full coupling between MIKE SHE and MIKE 11 has been developed. MIKE 11 computes water levels and flows in the river and floodplain system. The water levels, flows and flooded areas from MIKE 11 are then used as boundary conditions in MIKE SHE for calculation of the remaining parts of the hydrological cycle. The interactions between the river and the other components (aquifer, overland flow, etc) computed in MIKE SHE are then in turn transmitted back to MIKE 11. The coupling is illustrated in Fig. 2.

In the combined modelling system, the simulation takes place simultaneously in MIKE 11 and MIKE SHE. MIKE 11 simulates water levels and flows in rivers and on inundated areas. MIKE SHE simulates flow conditions in the remaining part of the hydrological cycle, with the water levels and area of distribution from MIKE 11 as the boundaries. The resulting inflows to the rivers and inundated areas are transferred to MIKE 11. During the simulation, a dynamic change in the MIKE 11 area of distribution takes place, depending on the extent of flooding. Numerically, the two systems may utilize different time steps. The data transfer between the two systems takes place through shared memory.

The MIKE SHE-MIKE 11 coupling is crucial for a correct description of the dynamics of the river-aquifer interaction. Firstly, the river width is larger than one MIKE SHE grid, in which case the MIKE SHE river-aquifer description is no longer valid. Secondly, the river/reservoir system comprises a large number of hydraulic structures, the operation of which cannot be accounted for in MIKE SHE. Thirdly, the very complex branch system with loops and flood cells needs a very efficient hydrodynamic formulation such as MIKE 11's.

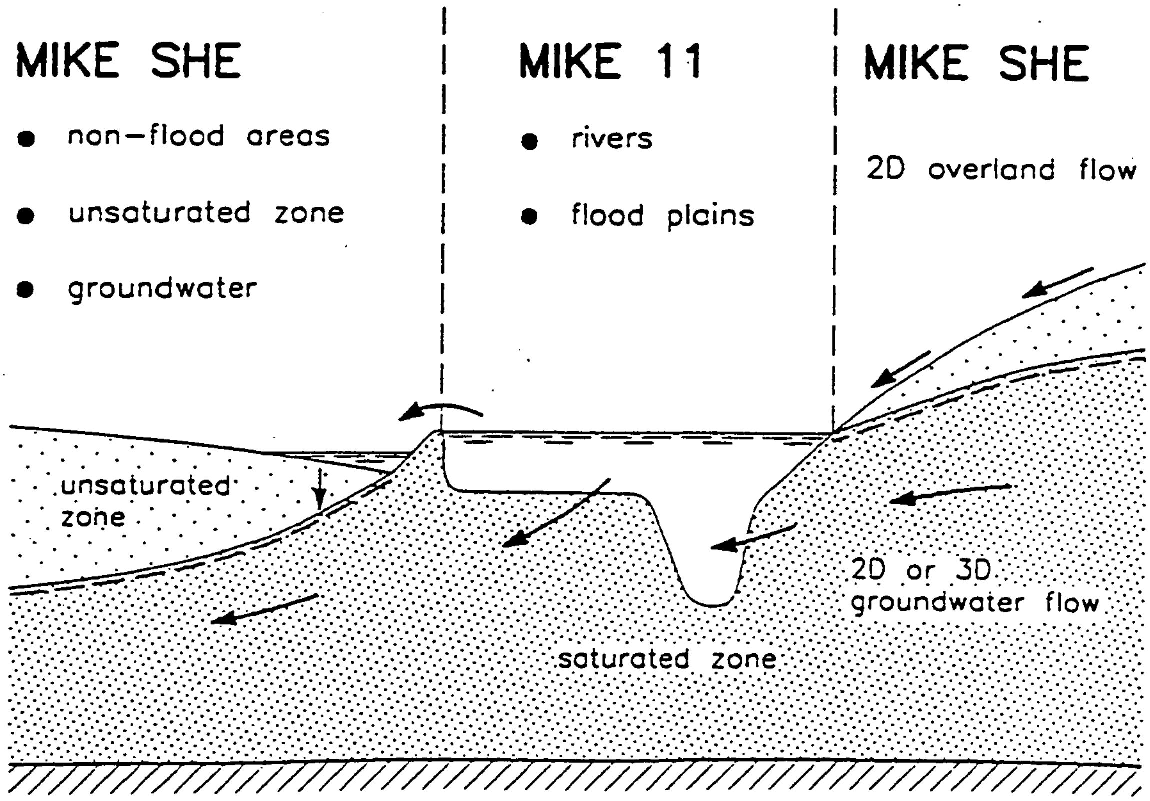

Fig. 2 Coupling of MIKE 11 and MIKE SHE modelling systems

4. Modelling studies in the Danubian Lowland - preliminary results

Comprehensive modelling studies are initiated and planned under the PHARE project [PHARE, 1992[9]; Refsgaard et al, 1994[10]]. In the present paper some preliminary results are presented with regard to ground water regime and hydrology of flood plains.

4.1 Ground water regime

The basic ground water flow modelling is a regional model covering an area of about 3.500 km^2 between Bratislava and Komarno. The main objectives of the regional model are to study the overall hydrological regime in the area and to provide reliable boundary conditions for the various sub-models.

The present ground water modelling is based on the data basis and previous modelling established by professor Igor Mucha's group, see e.g. [Mucha et al, 1992[11]]

Fig. 3 illustrates the different modelling applications and the scales on which modelling is carried out. Potential heads, varying in time and space, simulated with the regional model acts as boundary conditions for the sub models. The Danube recharges the groundwater throughout the year and hence the groundwater regime is, in most of the area, heavily influenced by the dynamics of the Danube.

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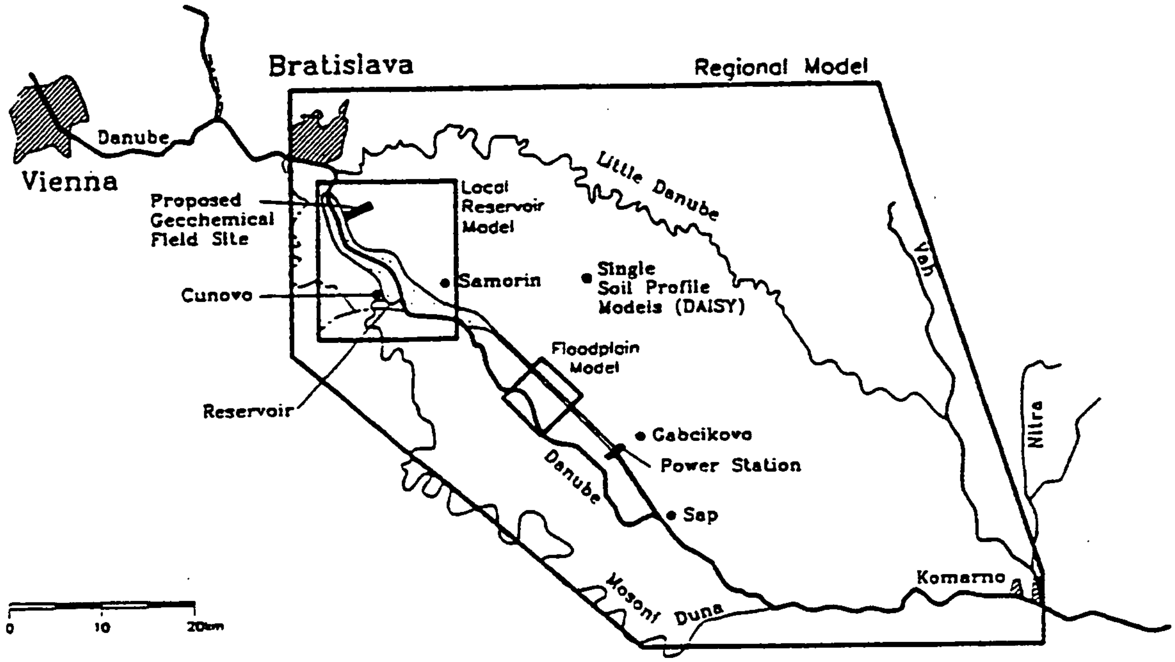

Fig. 3 Index map with project area and various scales for planned model applications.

Fig. 4 shows how the groundwater table, in three different distances from the river, responds to changes in the water level in the Danube. The results are from a preliminary model calibration for pre-dam conditions carried out in a 1000 m model grid and with a lumped description of the spatial variability of the soil properties in the unsaturated zone. The final calibration will be carried out for a two year period, in a 500 m model grid. The model will be validated by demonstrating its capability to reproduce observed groundwater levels after damming the Danube. The only model modification to be made in this context will be in the river geometry, reflecting the construction of the reservoir and the navigational canal.

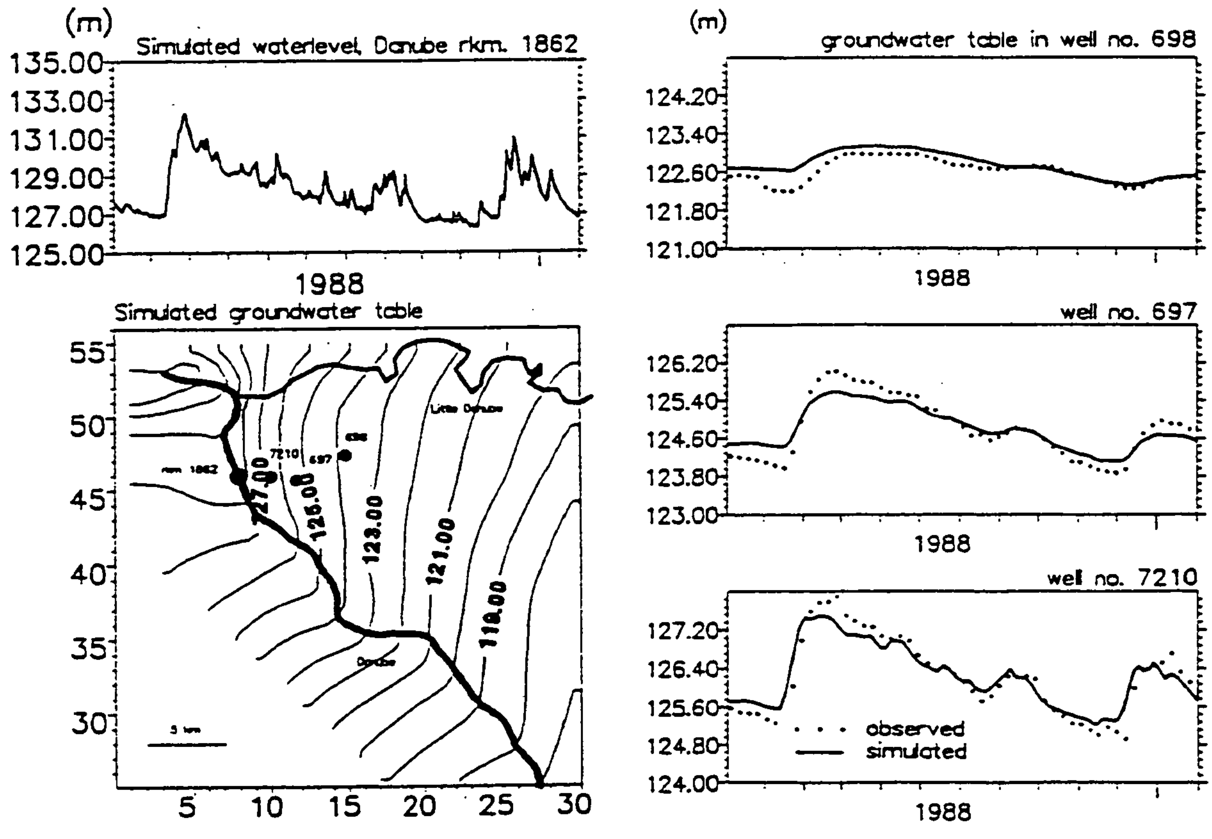

Fig. 4 Simulated and observed water tables for the regional model.

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4.2 Flood plain hydrology

The complexity of the floodplain with its river branch system is shown in Fig. 5. for the 20 km reach downstream the reservoir on the Slovakian side where alluvial forest occurs. In order to enable predictions of possible changes in floodplain ecology it is crucial to provide a detailed description of both the surface water and the groundwater systems in this area as well as of their interaction. For this purpose the MIKE SHE-MIKE 11 coupling is required.

The river branch system receives water from an inlet structure in the navigation canal at Dobrohost (see Fig. 5). This weir has a capacity of 234 m^3/s and together with the various hydraulic structures in the river branches, it controls the hydraulic, hydrological and ecological regime in the river branches and on the floodplains.

The floodplain model is a management tool for operation of the hydraulic structures, enabling an optimization of the hydraulic and ecological conditions for the unique floodplain environment. The floodplain model provides detailed information in time and space about water levels in river branches and on the floodplains, groundwater levels and soil moisture conditions in the unsaturated zone. Such information can directly be compared with quantitatively formulated ecological criteria.

Fig. 6 shows results from a preliminary model describing surface water interactions between floodplains and river branches. The preliminary model operates with a 50 m model resolution and is established for the downstream part of the river branch system as indicated in Fig. 5. Fig. 6 shows the centerline of the main river branches and the simulated inundated area at two different dates, June 11th and August 7th 1993, where the flow through the inlet weir was about 25 m^3/s and 15 m^3/s, respectively. Furthermore, the simulated water levels in a river branch and on the floodplain are shown.

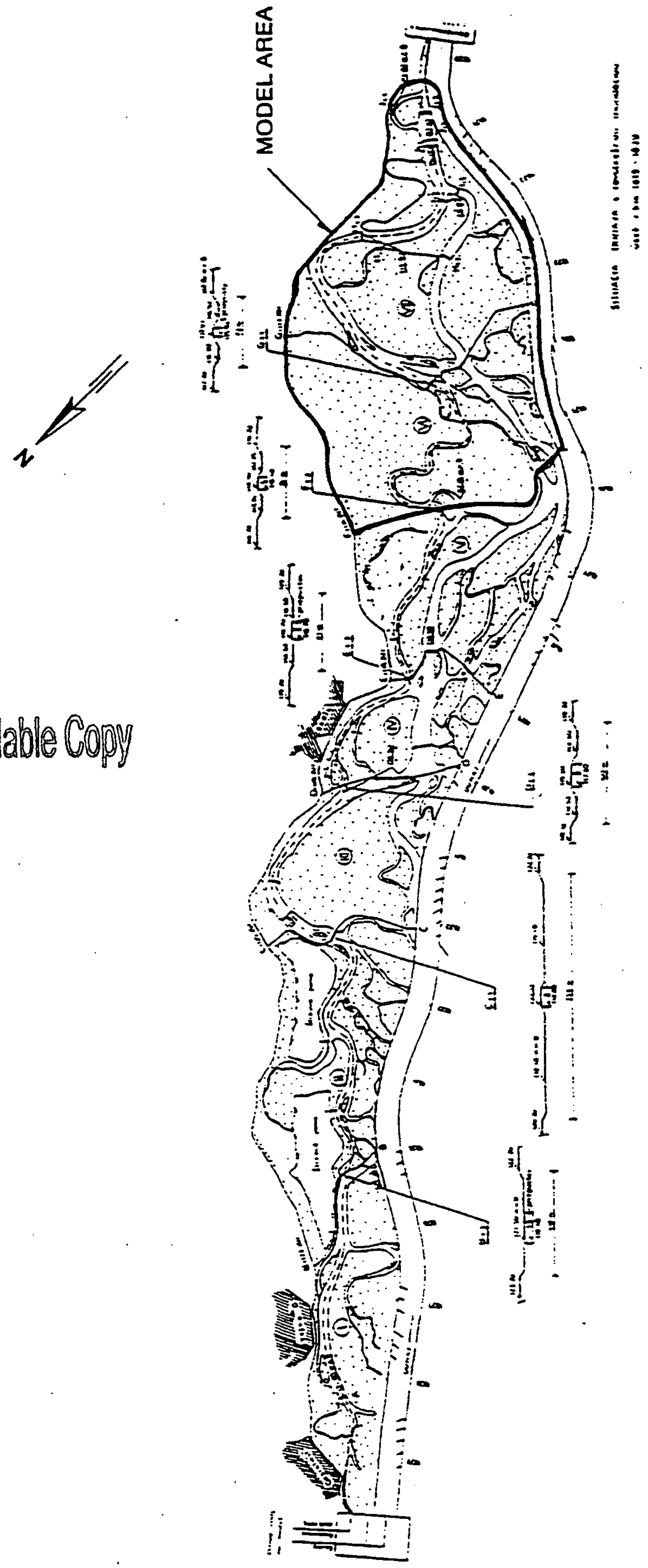

Fig. 5. Sketch of river branch system on the Slovakian floodplain for a reach of 20 km downstream the reservoir.

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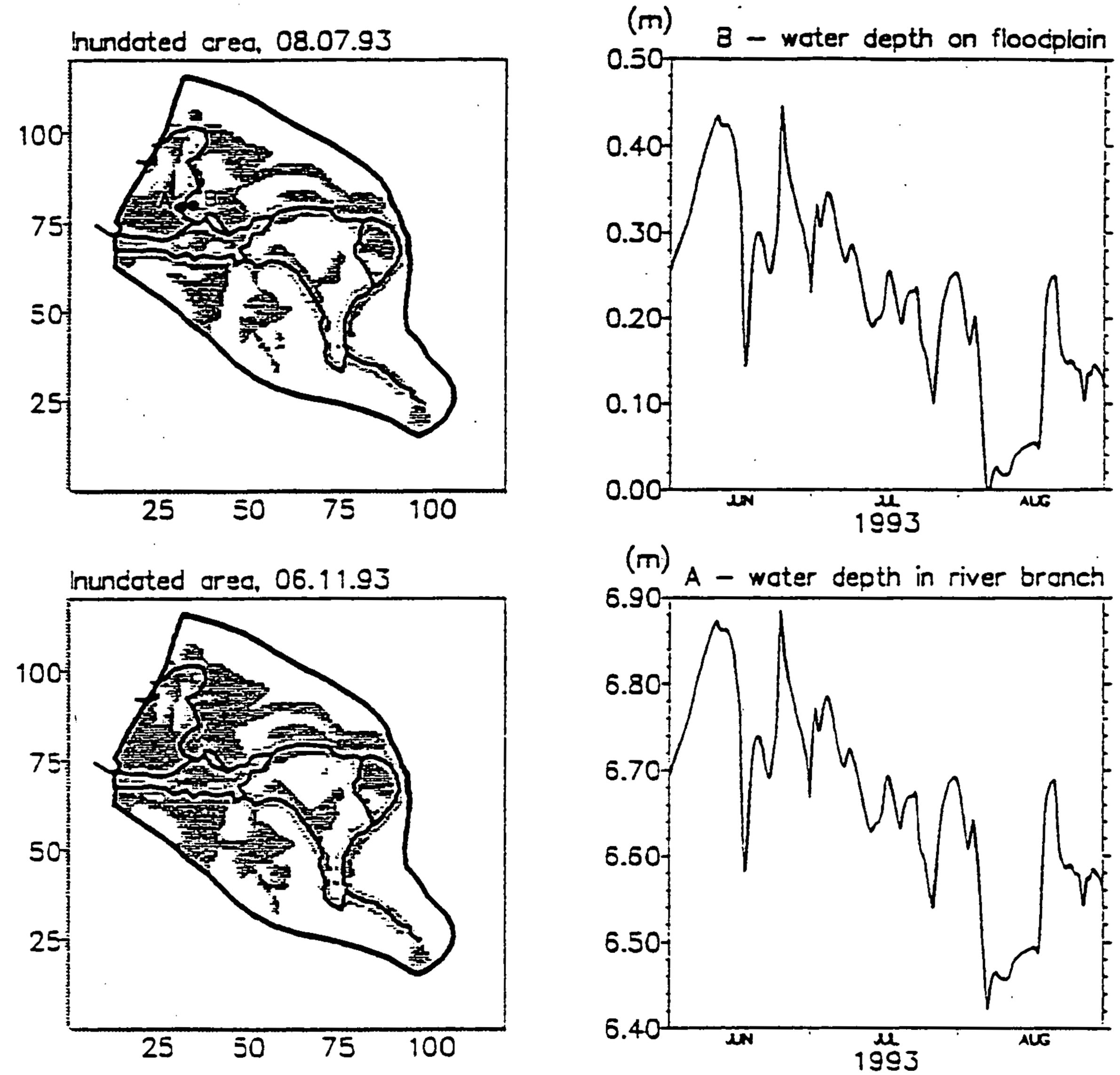

Fig. 6. Simulation results for a part of the Slovakian river branch system with a preliminary version of the floodplain model.

5. Conclusions

The ecological system of the Danubian Lowland is so complex with so many interactions between the surface and the subsurface water regimes and between physical, chemical and biological changes that a comprehensive mathematical modelling system is required in order to provide quantitative assessments of environmental impacts.

Such modelling system coupled with a comprehensive data base/GIS system is being developed under the PHARE project. When finally calibrated and verified this modelling and information system will provide the best available tool for providing assessments of the impacts on surface and ground water quantity and quality of alternative water management schemes.

In addition, the integrated system will enable detailed, quantitative predictions of surface and ground water regime in the floodplain area, including e.g. frequency, magnitude and duration of inundations. Such information constitutes a necessary basis for subsequent analysis of flora and fauna in the floodplain.

In the present paper some of the capabilities of the modelling system has been illustrated by preliminary results on ground water regime and flood plain hydrology.

ACKNOWLEDGEMENT

The PHARE project is being executed by the Slovak Ministry of the Environment.The Project Manager is Professor Igor Mucha from the Ground Water Consultants, Faculty of Natural Science, Comenius University (PRIF UK). A Danish-Dutch consortium of six organizations was selected as Consultant for the project. The Consultant is headed by Danish Hydraulic Institute (DHI) and comprises the following associated partners: DHV Consultants BV, The Netherlands; TNO-Applied Institute of Geoscience, The Netherlands; Water Quality Institute (VKI), Denmark; I Krüger Consult AS, Denmark; and the Royal Veterinary and Agricultural University, Denmark.

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An integrated eco and hydrodynamic model for prediction of wetland regime in the Danubian Lowland under alternative operation strategies for the Gabcikovo hydropower plant

by

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SYNOPSIS. An integrated mathematical modelling system describing flows, water quality processes, sediment transport/erosion in river, flood plain, reservoir and ground water system is being developed and fully coupled with a large data base and GIS system. The modelling and information system is being established in an ongoing project "Danubian Lowland - Ground Water Model". The system will be applied for assessing environmental impacts on eg ground water and floodplains of alternative water management strategies in the area, including alternative operation plans for the Gabcikovo hydropower scheme. The present paper gives a brief description of the established modelling and information system with special emphasis on the coupling between MIKE SHE and MIKE 11 and of the plans for model application during the project with special emphasis on aspects relating to wetland hydrology.

DANUBIAN LOWLAND - BACKGROUND

1. The Danubian Lowland between Bratislava and Komârno is an inland delta formed in the past by river sediments from the Danube. The entire area forms an alluvial aquifer, which throughout the year receives in the order of 25 m^3/s infiltration water from the Danube in the upper parts of the area and returns it into the Danube and the drainage channels in the downstream part. The aquifer is an important water resource for municipal and agricultural water supply.

2. Human influence has gradually changed the hydrological regime in the area. Construction of dams upstream of Bratislava together with exploitation of river sediments has significantly deepened the river bed and lowered the water level in the river. These changes have had a significant influence on the conditions of the ground water regime as well as the sensitive riverside forests downstream of Bratislava. In spite of this basically negative trend the floodplain area with its alluvial forests and the associated ecosystems still represents a very unique landscape of outstanding importance.

3. The construction of the hydraulic structures in connection with the hydropower plant at Gabcikovo also significantly affects the hydrological regime and the ecosystem of the region.

4. Industrial waste and municipal sewage from Bratislava and its surroundings together with the diffuse sources of agricultural fertilizers and agrochemicals are polluting the rivers, soil and ground water.

5. These physical and biochemical changes may reduce the atmospheric oxygen transport to the ground water and at the

same time increase the supply of organic matter which will change the oxidizing conditions to reducing conditions and thereby seriously deteriorate the ground water quality.

6. Due to the economical and ecological importance of the area comprehensive data collection programmes have been carried out for many years and a large number of studies have been made in the past. Some of the present environmental problems are published in (ref 1-2).

7. To utilize state-of-the-art modelling technology for addressing the water resources problems in the area the project "Danubian Lowland - Ground Water Model" has been defined within the PHARE programme agreed upon between the Commission of the European Communities and the Government of the Slovak Republic.

OBJECTIVE AND FRAMEWORK OF THE PHARE PROJECT

8. To understand and analyze the complex relationships between physical, chemical and biological changes in the surface- and subsurface water regimes requires multidisciplinary expertise in combination with advanced mathematical modelling techniques.

9. The overall project objective is to establish a reliable impact assessment model for the Danubian Lowland area, which enables the authorities to formulate optimal management strategies leading to a protection of the water resource and a sound ecological development for the area.

10. The PHARE project is being executed by the Slovak Ministry of the Environment. Specialists from the following Slovakian organisations are involved in various aspects of the project implementation:

- Comenius University, Faculty of Natural Science (PRIF UK)
- Water Research Institute (VUVH)
- Irrigation Research Institute (VUZH)

A Danish-Dutch consortium of six organisations was selected as consultant for the project.

11. The project was initiated in the beginning of 1992 and will be completed by the end of 1995. At present the necessary equipment has been delivered to the project in Bratislava and the modelling and information system has been installed on computer workstations. The modelling work has been initiated; however no final model calibrations nor predictions have been carried out at this stage.

ESTABLISHMENT OF DANUBIAN LOWLAND INFORMATION SYSTEM (DLIS)

12. An automated system is presently being developed to support the modelling activities. The integrated modelling system will be interfaced to a central information system. The central information system, called Danubian Lowland Information System (DLIS), will provide the different models with the necessary data and functionality to elaborate further on the modelling results.

13. Because of the complexity and the amount of data involved - first estimates indicate about 2 Gbyte of data - major attention has been paid to the development of the underlying data model. The information needed for the modelling originates from different monitoring networks, which are maintained and observed by different institutes. The larger part of the data is available from automated archives and can be loaded into the system from magnetic medium.

14. The information to be incorporated in DLIS has a pronounced spatial character. The two main components of the GIS are a geographical information system (GIS), for which ARC/INFO has been selected, and a relational data base management system (RDBMS), for which INFORMIX has been selected.

ESTABLISHMENT OF INTEGRATED MODELLING SYSTEM

15. The integrated modelling tool, which will form the basis for all the modelling activities, is based on the following packages which can be used individually or brought together in an integrated manner:

* **MIKE SHE** which, on catchment scale, can simulate the major flow and transport processes of the hydrological cycle which are traditionally divided in separate components:
  - 1-D flow and transport in the unsaturated zone
  - 3-D flow and transport in the ground water zone
  - 2-D flow and transport on the ground surface
  - 1-D flow and transport in the river.

  All the above processes are fully coupled allowing for feedbacks and interactions between components. In addition to the above mentioned components, MIKE SHE includes modules for multi-component chemical reactions in the unsaturated and saturated zone as well as a component for oxygen consumption and transport in the unsaturated zone.
* **MIKE 11**, which is a one-dimensional river modelling system. MIKE 11 is used for hydraulics, sediment transport and morphology, and water quality. The modules for sediment transport and morphology are able to deal with cohesive and non-cohesive sediment transport, as well as the accompanying morphological changes of the river bed. The non-cohesive model will operate on a number of different grain sizes, taking into account shielding effects. The cohesive model deals both with consolidation of the river bed and flocculation.
* **MIKE 21**, which is a two-dimensional hydrodynamic modelling system. MIKE 21 is used for reservoir modelling, including hydrodynamics, sediment transport and water quality. The sediment transport modules deals with both cohesive and non-cohesive sediment, and the non-cohesive module will operate on a number of different grain size fractions.
* Both of the above mentioned models include **River/Reservoir Water Quality (WQ) and Eutrophication (EU)** modules to describe oxygen, ammonium, nitrate and phosphorus concentrations and oxygen demands as well as eutrophication issues.
* **DAISY** is a one-dimensional root zone model for simulation of crop production, soil water dynamics, and nitrogen dynamics in crop production for various agricultural management practices and strategies. The particular processes considered include transformation and transport involving water, heat, carbon and nitrogen.

16. The above mentioned models are all generalized tools with comprehensive applicability ranges, and they are well proven in a large number of international projects. In addition, some model modifications are being carried out during the project in order to accommodate the very special

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environment and problems observed in the area.

17. With regard to simulation of floodplain hydrology and ecology the core of the integrated modelling system is constituted by the MIKE SHE, the MIKE 11 and a newly developed, full coupling of the two systems as described in the following three sections.

MIKE SHE

18. The European Hydrological System - SHE was developed in a joint effort by Institute of Hydrology (UK), SOGREAH (France) and Danish Hydraulic Institute. It is a deterministic, fully-distributed and physically-based modelling system for describing the major flow processes of the entire land phase of the hydrological cycle. A description of the SHE is given in (Ref. 3-4). Since 1987 the SHE has been further developed independently by the three respective organizations which now are University of Newcastle (UK), Laboratoire d'Hydraulique de France and DHI. DHI's version of the SHE, known as the MIKE SHE, represents significant new developments with respect to user interface, computational efficiency and process descriptions.

19. MIKE SHE solves the partial differential equations for the processes of overland and channel flow, unsaturated and saturated subsurface flow. The model is completed by a description of the processes of snow melt, interception and evapotranspiration. The flow equations are solved numerically using finite difference methods.

20. In the horizontal plane the catchment is discretized in a network of grid squares. The river system is assumed to run along the boundaries of these. Within each square the soil profile is described in a number of nodes, which above the groundwater table may become partly saturated. Lateral subsurface flow is only considered in the saturated part of the profile. Fig. 1 illustrates the structure of the MIKE SHE. A description of the methodology and some experiences of model application are presented in (Ref. 5-8).

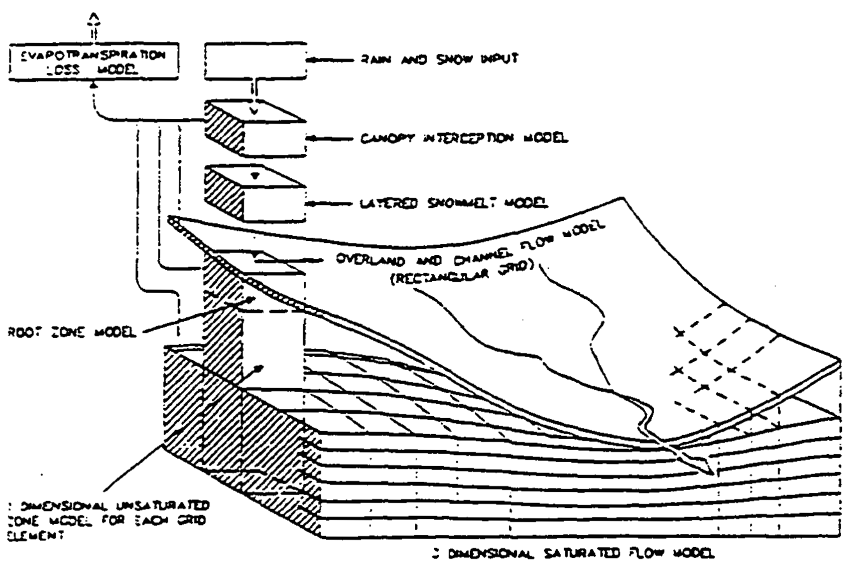

Fig. 1. Schematic presentation of the MIKE SHE.

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21. The spatial and temporal variations in the catchment characteristics and meteorological input are provided in a series of two-dimensional matrices of grid square codes. To each code is further attached a number of attributes describing either parametric data or input data.

22. The distributed description in the MIKE SHE allows the user to include and test against spatially varying data. MIKE SHE is a multi output model which besides discharge in any river link also produces information about e.g. water table elevations, soil moisture contents, infiltration rates, evapotranspiration, etc. in each grid square.

MIKE 11

23. MIKE 11 is a comprehensive, one-dimensional modelling system for the simulation of flows, sediment transport and water quality in estuaries, rivers, irrigation systems and other water bodies. It is a 4th generation modelling package designed for micro-computers with DOS or UNIX operating systems and provides the user with an efficient interactive menu and graphical support system with logical and systematic layouts and sequencing in the menus. The package was introduced in 1989 and today the number of installations world-wide exceeds 300. The modular structure of MIKE 11 is illustrated in Fig. 2.

24. The hydrodynamic module of MIKE 11 is based on the complete partial differential equations of open channel flow (Saint Venant). The equations are solved by implicit, finite difference techniques. The formulations can be applied to branched and looped networks and quasi two-dimensional flow simulations on floodplains.

25. MIKE 11 operates on the basis of information about the river and the floodplain topography, including man-made hydraulic structures such as embankments, weirs, gates, dredging schemes and flood retention basins. The hydrodynamic module forms the basis for morphological and water quality studies by means of add-on modules.

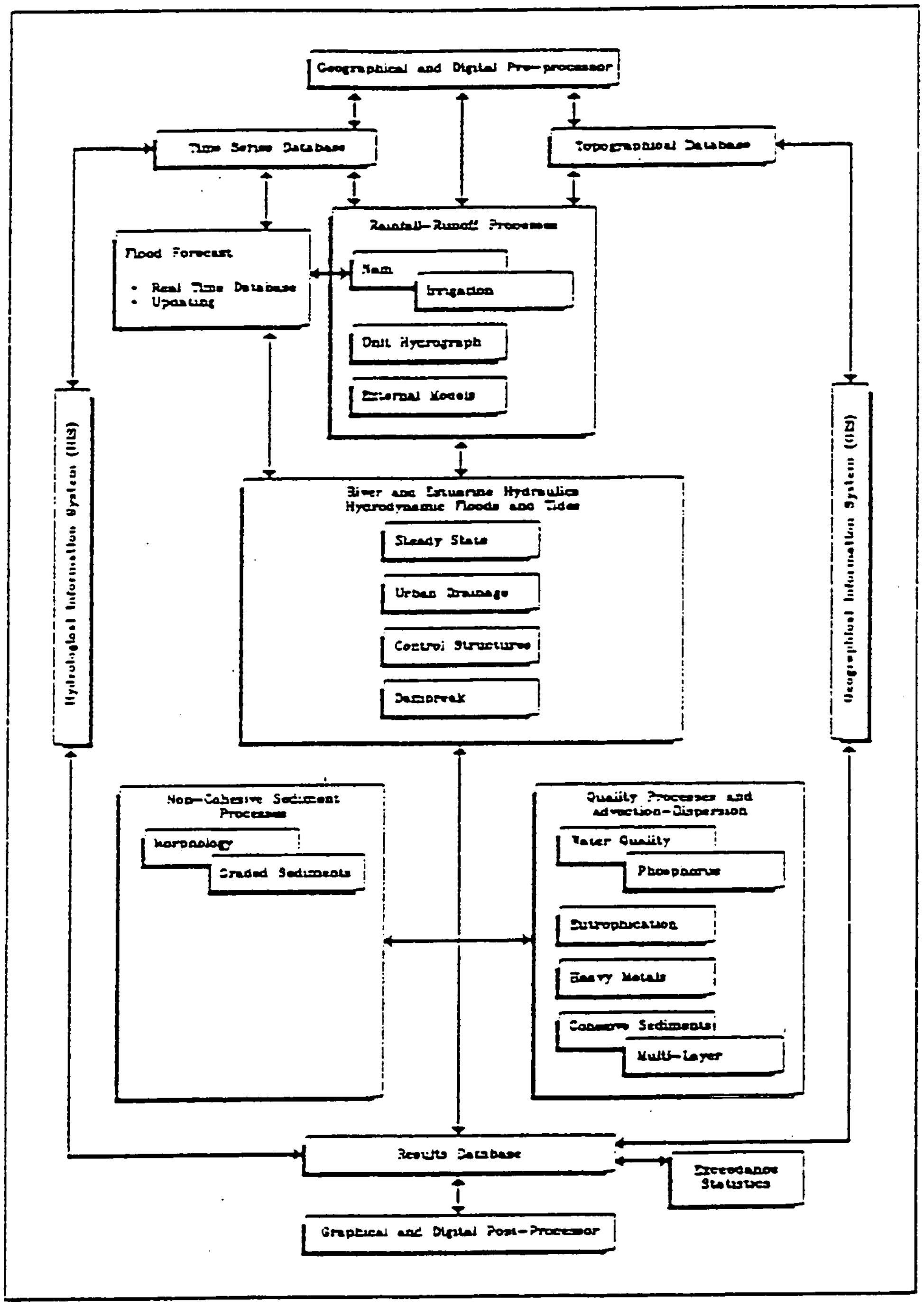

Fig. 2. Modular structure of MIKE 11

A COUPLING OF MIKE SHE AND MIKE 11

26. The focus in MIKE SHE lies on catchment processes with a comparatively less advanced description of river processes. In contrary MIKE 11 has a more advanced description of river processes and a simpler catchment description than MIKE SHE. Hence, for cases where full emphasis is needed for both river and catchment processes a coupling of the two modelling systems is required.

27. A full coupling between MIKE SHE and MIKE 11 has been developed. MIKE 11 computes water levels and flows in the river and floodplain system. The water levels, flows and flooded areas from MIKE 11 are then used as boundary conditions in MIKE SHE for calculation of the remaining parts of the hydrological cycle. The interactions between the river and the other components (aquifer, overland flow, etc) computed in MIKE SHE are then in turn transmitted back to MIKE 11.

28. The two systems are run simultaneously with full exchange of data. Numerically, the two systems may utilize different time steps. The data transfer between the two systems takes place through shared memory.

29. The MIKE SHE-MIKE 11 coupling is crucial for a correct description of the dynamics of the river-aquifer interaction. Firstly, the river width is larger than one MIKE SHE grid, in which case the MIKE SHE river-aquifer description is no longer valid. Secondly, the river/reservoir system comprises a large number of hydraulic structures, the operation of which cannot be accounted for in MIKE SHE. Thirdly, the very complex branch system with loops and flood cells needs a very efficient hydrodynamic formulation such as MIKE 11's.

30. The complexity of the floodplain with its river branch system is shown in Fig. 3. for the 20 km reach downstream the reservoir on the Slovakian side where alluvial forest occurs. In order to enable predictions of possible changes in floodplain ecology it is crucial to provide a detailed description of both the surface water and the groundwater systems in this area as well as of their interaction. For this purpose the MIKE SHE-MIKE 11 coupling is required.

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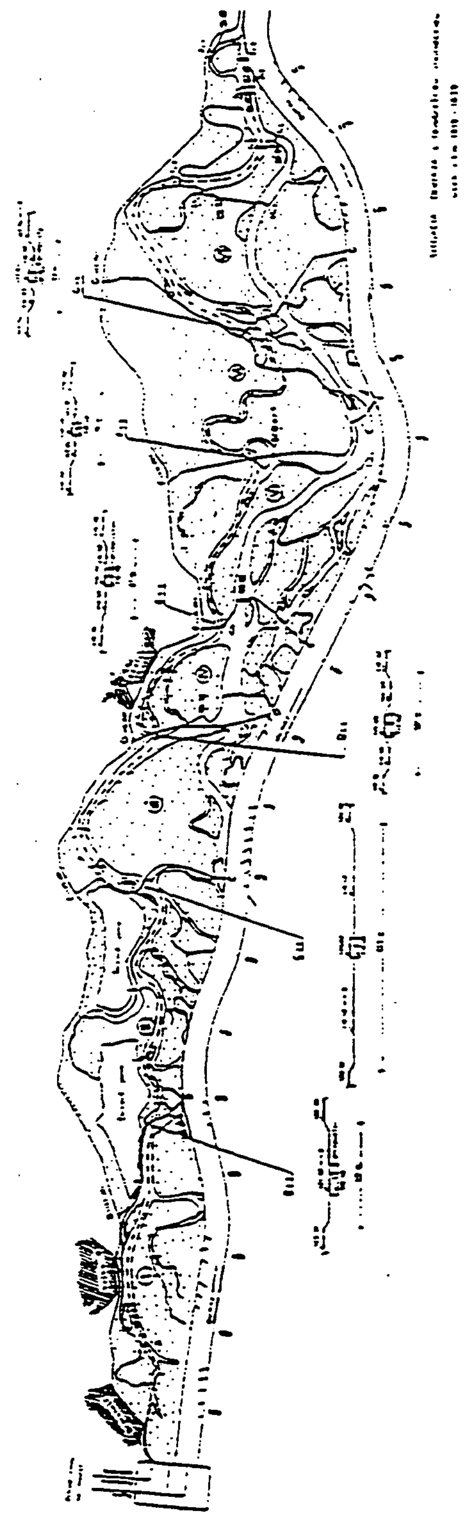

Fig. 3. Sketch of river branch system on the Slovakian floodplain for a reach of 20 km downstream the reservoir.

MODELLING STUDIES IN THE DANUBIAN LOWLAND

31. The modelling studies initiated and planned under the PHARE project involve a number of disciplines and processes with different space and time scales as outlined below. An index map also illustrating the different spatial scales is shown in Fig. 4.

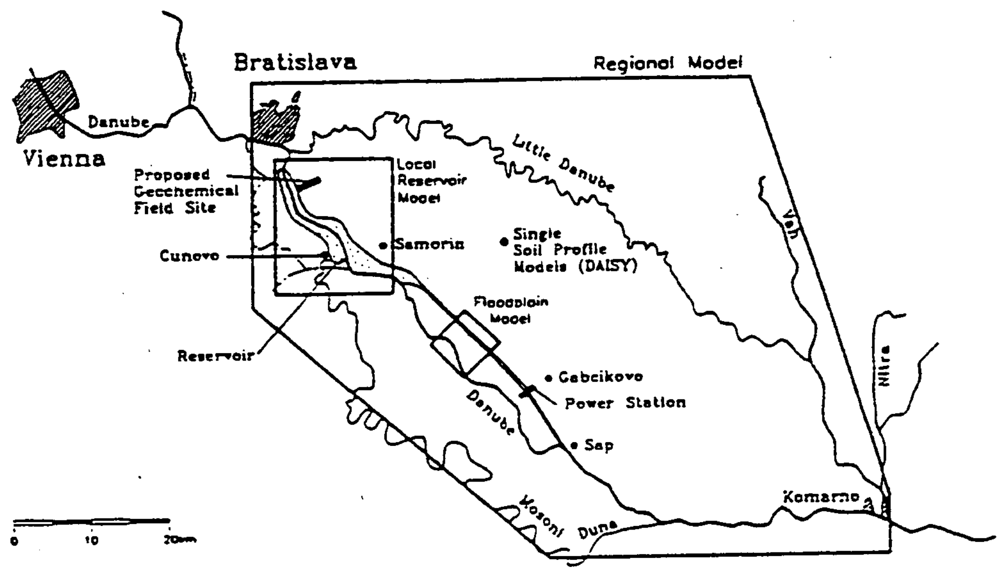

Fig. 4. Index map of the Danubian Lowland with indications of various spatial modelling scales.

32. River and Reservoir Flow and Sediment Transport. For the simulations of flows and sediment transport in the reservoir and the old Danube a combination of one and two-dimensional numerical models is applied.

33. MIKE 11 is being established for the Danube from a point within Austria to Komárno. It includes possibilities for imposing specific operation of the structures in connection with the hydropower plant at Gabcikovo and the reservoir weir at Cunovo. For the old Danube reach between Cunovo and Sap the complex channel branch system with all its internal regulating structures is included together with possibilities for directly describing inundation depths and coverage of the flood plains in between the branches. Thus the model is able to describe both low and high flow conditions as well as all possible regulation possibilities.

34. For the regional ground water studies the Little Danube and the irrigation and drainage systems are also included in the model setup.

35. The model will be calibrated on conditions from the 1960's as well as on the present conditions.

36. Long term morphological simulations will be carried out in order to assess bed level changes and composition of sediment in the backwater zone of the reservoir and in the old Danube (due to e.g. flushing of sediment from the reservoir and possible dumping of dredged material).

37. Hydrodynamic modelling with MIKE 21 has already been carried out for different options of reservoir alignments and deflecting structure designs, see Fig 5.

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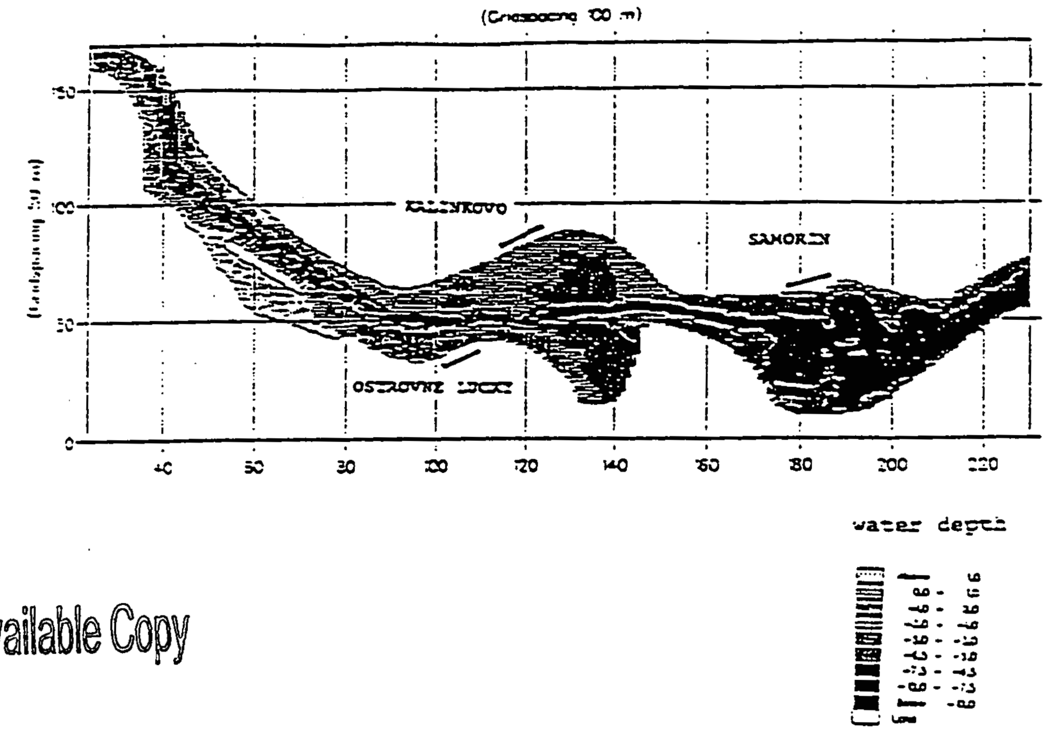

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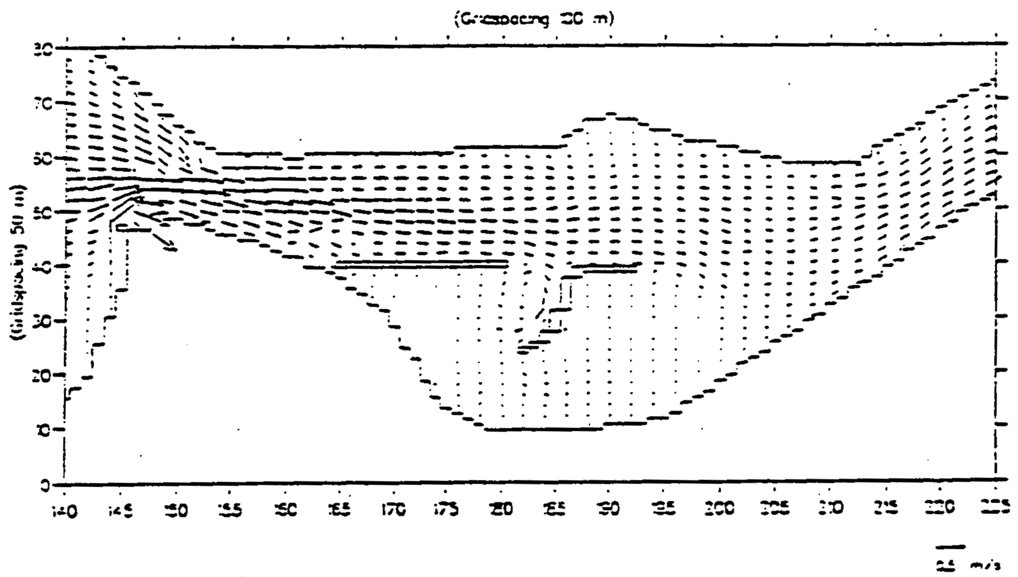

Fig. 5. Example of MIKE 21 model prediction of flow pattern in the Samorin reservoir with the designed reservoir alignment and deflecting structures. On the upper figure the bathymetry for the entire reservoir is shown together with the location of three major well fields for water supply. The lower figure shows the simulated velocities corresponding to a discharge of 1400 m3/s into the reservoir and power canal.

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38. The sediment transport modelling of suspended load will include both cohesive and non-cohesive transport. Also resuspension during flood of sediment deposited in periods with low flow will be accounted for. The boundary conditions in terms of flux and sediment transport at the upstream boundary, water level/flux at the entrance to the power channel, water level/flux at the weir will be provided from the MIKE 11 simulations. Sediment transport boundary conditions will consist of grain size distributions and suspended sediment concentration for each fraction.

39. The predictions will provide information about flows, water levels, grain size distributions and depths of the deposited suspended sediments in the reservoir. These results will be used both in the surface water quality modelling as well as in the ground water quality and quantity modelling. For assessment of the morphological consequences of different flushing schemes, the MIKE 21 non-cohesive sediment transport module will be applied.

40. Surface Water Quality. In order to highlight the oxygen status of the surface water, particularly during low flow situations and in slow flowing branches of the Danube, MIKE 11WQ is applied. The MIKE 11WQ describes the diurnal variation in the water quality parameters, i.e. the concentration of organic matter, oxygen, ammonium and nitrate in the water. The diurnal variation is especially important in the branches of the old Danube with a significant macrovegetation, in areas where severe eutrophication occurs and in periods with relatively low flow velocities.

41. The model will be used both in the old Danube with its branches as well as in the reservoir (here as a submodel to MIKE 21). Output from the eutrophication model, (which describes the daily average production), with respect to levels of oxygen production from primary producers, can be used in these calculations.

42. A description of the horizontal differences in the algae growth and the possible sedimentation of organic matter in the reservoir will be carried out by using the two-dimensional MIKE 21EU. Because of the relatively shallow areas in the reservoir macrophytes should be included in the future eutrophication modelling.

43. The eutrophication effect in the old Danube and the branch system will be most severe in periods with low flow. In these periods the system can be described by a branched 1-D system because the old Danube will be described by a branched/looped MIKE 11 model. The eutrophication effects can also be described by the one dimensional MIKE 11EU-model including macrophytes.

44. Ground water flow modelling. The application of MIKE SHE at three spatial scales, see Fig. 3, will support different types of modelling studies:

- **Regional scale.** The aim at this scale is to provide a framework for regional predictions and provide realistic boundary conditions (usually head boundaries) for local models. Some management options could have subregional or even regional implications requiring a reliable model on this scale. For the purposes of detailed modelling in local areas, where a very detailed description is required, it is only necessary and only technically feasible to establish a model for a smaller area. If the exact boundary conditions are not easily established, the regional model can provide the dynamic boundary conditions

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which may account for the conditions outside the model area.

The regional model includes the entire Zitny Ostrov area and cover approximately 3000 km². The overall hydrological regime will be simulated taking into account all the major surface water systems.

* **Local scale.** For studies of the local conditions MIKE SHE will be set up in small areas to describe the flow and transport conditions with a high degree of detail both in horizontal and vertical directions (a fully 3-D description). These models will serve as a basis for the detailed description of different aspects, e.g.:
  - Geochemical processes around the reservoir area; and
  - ecological effects in the Danubian flood plain area.
* **Transect/Plot scale.** Model simulations on this scale will basically serve to study specific processes which either require a very fine spatial resolution or which can be described by one-dimensional flows, e.g.:
  - Geochemical processes (e.g. along a transect in connection with field investigations);
  - flow and solute (e.g. nitrate) processes in the unsaturated zone (soil columns); and
  - analysis of oxygen transport from the atmosphere through the unsaturated zone to the water table (transect or soil column).

45. Geochemical modelling. The hydrogeochemical modelling focuses on the part of the aquifer system in the vicinity (2-3 km) of the Danube river.

46. The infiltration of oxygen and nitrate through river (and reservoir) beds with different compositions, e.g. fine sediments rich in organic material and gravel sediments with a small amount of organic material, will result in different supplies of oxidation capacity to the aquifer system at the different river-aquifer interfaces. In some parts of the interface between river and aquifer, water with a low oxidation capacity and perhaps even anoxic will infiltrate due to the build up of a low permeable layer of fine sediment in the river after the completion of the reservoir. It is therefore important to handle the transfer of oxidation capacity through various river bed systems correctly in order to be able to characterize the supply of oxidation capacity to the aquifer system. Oxidation capacity can also be added to the system through infiltration from the unsaturated zone. Further, the effects of a fluctuating water table in the riverine area in bringing oxygen (and nitrate) to the aquifer will be studied. 47. The hydrogeochemical modelling will focus on the oxidation/reduction processes in the river bed and aquifer systems. The total amount of oxygen and nitrate is equal to the oxidation capacity added to the systems through the river and unsaturated zone (SO_4-reduction has not been considered here). Bulk organic matter, either in dissolved or solid form, is the reduction capacity. The hydrogeochemical situation will be that of a reduction of oxygen and nitrate by kinetically-controlled oxidation of organic matter. Oxygen must be consumed first before nitrate is reduced.

48. Unsaturated zone and agrochemical modelling. The effect of the reservoir on the productivity in the Zitny Ostrov is of direct concern. From the calibrated regional

model the present and future ground water levels will be simulated and the area can be classified according to its water supply for crops in the growing season.

49. Combining these predictions with the DAISY agricultural model the productivity and the irrigation needs before and after implementation of a given management scenario for the dam can be simulated for selected classes of water table depths and crops.

50. Although the amounts of nitrate leached are not extremely large, they are a problem for the general quality of the ground water. The losses can be reduced through changes in amounts of N applied, timing of application, optimal use of manure, and by optimal irrigation practices. Different scenarios can be analyzed through modelling.

51. Simulation with DAISY (Ref.9) can provide estimates of former and future nitrate loads leaching from the root zone under different conditions and to map "leaching hazard". Through discussions with the relevant agricultural institutions a number of scenarios can be defined with improved agricultural systems/protected areas and they can analyze how this would influence the leaching losses and ground water quality.

52. Modelling Ecological Effects in the Flood Plain. The ecological functioning of the floodplain is governed by the dynamics of inundation, flushing and ground water level fluctuations. These factors will form part of the modelling of the floodplain area.

53. The MIKE SHE/MIKE 11 model will be set up for an area which forms part of the existing monitoring system (for biomonitoring and forestry).

54. The model will be given sufficient detail in order to simulate the inundation and flushing regime at various discharges in the old Danube in order to predict changes in ecotype diversity. The horizontal model discretization is envisaged to be in the order of 50 m. The model will also allow the prediction of ground water levels, soil moisture regime of the floodplain in relation to channel and river branch development (e.g. morphology and sedimentation).

55. Water quality aspects will be considered as being included using MIKE 11 WQ/EU, thus enabling predictions of the water quality and eventually macrophyte growth.

CONCLUSIONS

56. The ecological system of the Danubian Lowland is so complex with so many interactions between the surface and the subsurface water regimes and between physical, chemical and biological changes that a comprehensive mathematical modelling system is required in order to provide quantitative assessments of environmental impacts.

57. Such modelling system coupled with a comprehensive data base/GIS system is being developed under the PHARE project. When finally calibrated and verified this modelling and information system will provide the best available tool for providing assessments of the impacts on surface and ground water quantity and quality of alternative water management schemes.

58. In addition, the integrated system will enable detailed, quantitive predictions of surface and ground water regime in the floodplain area, including e.g. frequency, magnitude and duration of inundations. Such information constitutes a necessary basis for subsequent analysis of flora and fauna in the floodplain.

ACKNOWLEDGEMENT

59. The PHARE project is being executed by the Slovak Ministry of the Environment.The Project Manager is Professor Igor Mucha from the Ground Water Consultants, Faculty of Natural Science, Comenius University (PRIF UK). A Danish-Dutch consortium of six organizations was selected as Consultant for the project. The Consultant is headed by Danish Hydraulic Institute (DHI) and comprises the following associated partners: DHV Consultants BV, The Netherlands; TNO-Applied Institute of Geoscience, The Netherlands; Water Quality Institute (VKI), Denmark; I Krüger Consult AS, Denmark; and the Royal Veterinary and Agricultural University, Denmark. The author of the present paper is the Team Leader for the Consultant.

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ENGINEERING PROPERTIES OF VEGETATION 2

M. E. Styczen and R. P. C. Morgan

2.1 INTRODUCTION

Vegetation provides a protective layer or buffer between the atmosphere and the soil. Through the hydrological cycle, it affects the transfer of water from the atmosphere to the earth's surface, soil and underlying rock. It therefore influences the volume of water contained in rivers, lakes, the soil and groundwater reserves. The above-ground components of the vegetation, such as leaves and stems, partially absorb the energy of the erosive agents of water and wind, so that less is directed at the soil, whilst the below-ground components, comprising the rooting system, contribute to the mechanical strength of the soil.

Traditionally, the role of vegetation has been viewed rather simplistically, as seen by the somewhat superficial way it is dealt with in water erosion studies. The most commonly used approach has been to assign to it a coefficient, such as the C-factor in the Universal Soil Loss Equation (Wischmeier and Smith, 1978) which, for a certain stage of growth and plant density, describes the ratio of soil loss when vegetation is present to the amount lost on a bare soil. Values of this soil loss ratio are derived experimentally from field trials and, while they are true values for those situations, they cannot be readily used to predict the effect of the same or other vegetation in different climatic and pedological conditions.

Wischmeier (1975) tried to split the C-factor into C_I, C_{II} and C_{III} subfactors (Figure 2.1). C_I describes the effect of the presence of a plant canopy at some elevation above the soil. C_{II} is defined as the effect of a mulch or close-growing vegetation in direct contact with the soil surface. Root effects are not included. C_{III} represents the residual effects of land use on soil structure, organic matter content and soil density, the effects of tillage or lack of tillage on surface roughness and soil porosity, and the effects of roots, subsurface stems and biological activity in the soil. This approach has been used in erosion prediction models (Beasley, Huggins and Monke, 1980; Park, 1981; Park, Mitchell and Scarborough, 1982) but is limiting because at least two of the three subfactors may influence more than one erosion process. It is difficult therefore to give them a precise physical meaning.

The conflicting views, provided by field and laboratory experiments (Figure 2.2) on what level of vegetation cover is required to reduce the soil loss ratio from 1.0 to 0.5, illustrate the inadequacy of the above approach. In order to understand the role of vegetation in combating erosion it is necessary to:

1. understand the erosion processes;
2. consider how each of these processes may be affected by vegetation;
3. determine the salient properties of the vegetation which most affect these processes;
4. try to quantify the combined effect of vegetation on the processes acting together in different situations.

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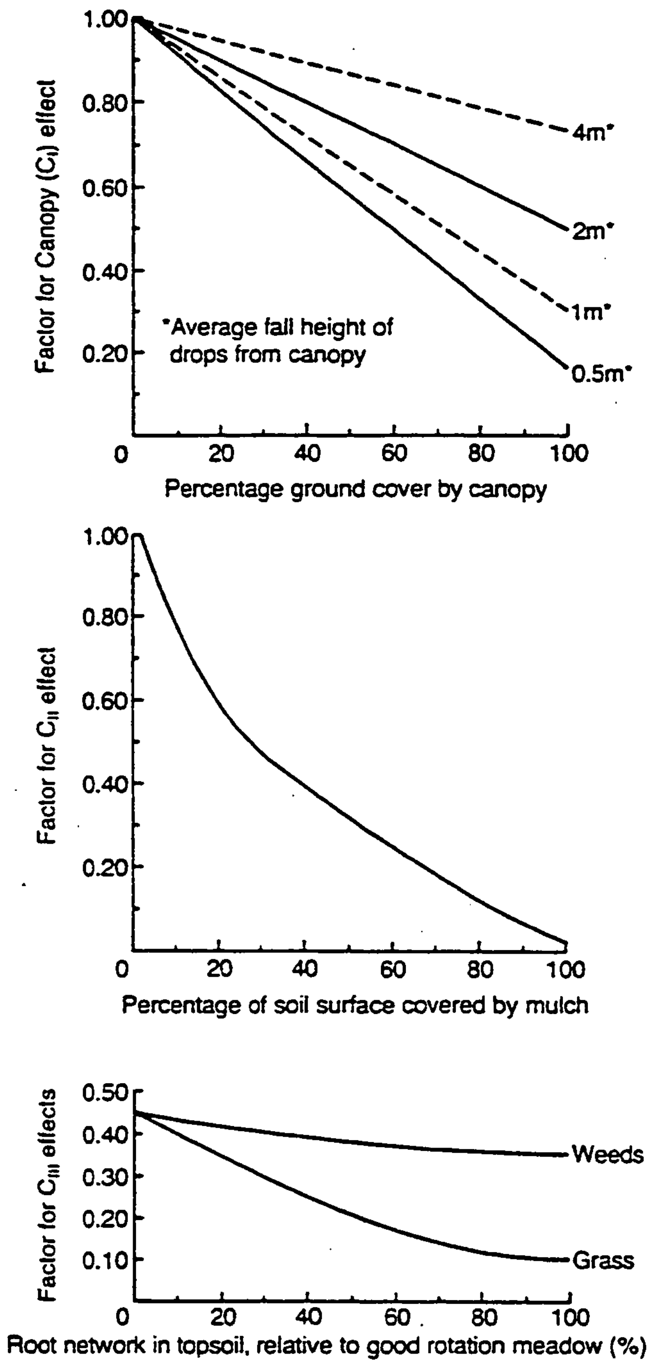

Figure 2.1 Soil loss ratios for subfactors of the C-factor in the Universal Soil Loss Equation (after Wischmeier, 1975). C_I describes the canopy effect, C_{II} the effect of plant residues and ground vegetation, and C_{III} the residual effects of previous land use. The graph shown here for the C_{III} effect applies to previously undisturbed land only and not to cropland or construction sites.

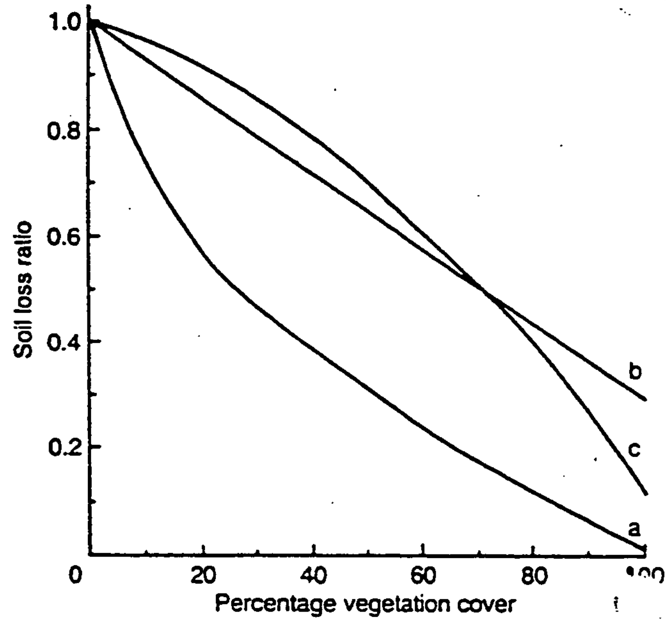

Figure 2.2 Examples of relationships between the soil loss ratio and percentage vegetation cover. a = ground level vegetation (Laflen and Colvin, 1981); b = vegetation canopy at 1 m above the ground (Wischmeier, 1975); c = oat straw mulch (Singer and Blackard, 1978).

Such a detailed understanding is difficult to achieve. It is hampered by the fact that much previous research has concentrated on establishing C-factor values rather than on understanding how vegetation operates within the erosion system. Analysis is also hampered by the complexity of the interaction between vegetation, climate, soil properties and hydrology. Nevertheless, the relatively low rates of erosion observed in well-vegetated areas compared with the catastrophic rates which can arise when vegetation is cleared, demonstrate that vegetation performs a major engineering role in protecting the landscape. This chapter aims to explore that role by reviewing its hydrological, hydraulic and mechanical effects. These are summarized diagrammatically in Figure 2.3.

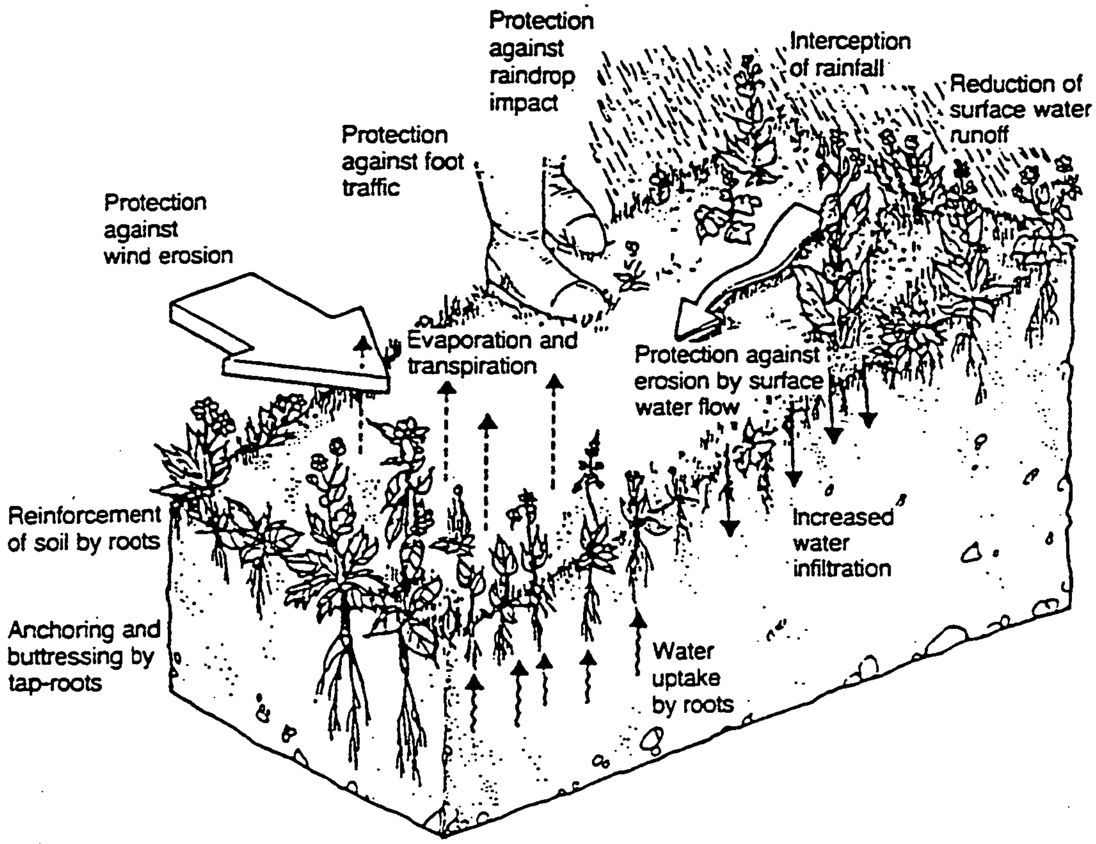

Figure 2.3 Engineering role of vegetation (after Coppin and Richards, 1990).

2.2 HYDROLOGICAL EFFECTS OF VEGETATION

2.2.1 EVAPOTRANSPIRATION

Evapotranspiration is the combined process of the removal of moisture from the earth's surface by evaporation and transpiration from the vegetation cover. Evapotranspiration from plant surfaces is compared to the equivalent evaporation from an open water body. The two rates are not the same because the energy balances of the surfaces are markedly different. For example, the albedo value, defined as the proportion of incoming short-wave radiation which is reflected, is, depending on the altitude of the sun, about 0.1 for water, but varies between 0.1 and 0.3 for a plant cover. The effect of vegetation is expressed by the E_t/E_o ratio, where E_t is the evapotranspiration rate for the vegetation cover and E_o is the evaporation rate for open water. Table 2.1 gives some typical values for plant covers at different stages of growth and in different seasons (Withers and Vipond, 1974; Doorenbos and Pruitt, 1977).

The values of E_t/E_o ratios assume that evapotranspiration is not limited by the supply of water; in other words, it is taking place at the potential rate (E_p). Where high rates of evapotranspiration occur, however, the top layers of the soil rapidly dry out and the plants find it more difficult to extract water from the soil by suction through the roots. To prevent dehydration, plants reduce their transpiration so that actual evapotranspiration (E_a) becomes less than potential. The ratio of actual to potential evapotranspiration (E_a/E_p) depends upon the soil moisture deficit (SMD) which is defined as the difference between the reduced soil moisture level and that pertaining at field capacity. The

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Table 2.1 E_t/E_o ratios for selected plant covers (after Withers and Vipond, 1974; Doorenbos and Pruitt, 1977)

Plant (crop) cover	E_t/E_o *ratio*
Wet (padi) rice	1.35
Wheat	0.59–0.61
Maize	0.67–0.70
Barley	0.56–0.60
Millet/sorghum	0.62
Potato	0.70–0.80
Beans	0.62–0.69
Groundnut	0.50–0.87
Cabbage/Brussels sprouts	0.45–0.70
Banana	0.70–0.77
Tea	0.85–1.00
Coffee	0.50–1.00
Cocoa	1.00
Sugar cane	0.68–0.80
Sugar beet	0.73–0.75
Rubber	0.90
Oil palm	1.20
Cotton	0.63–0.69
Cultivated grass	0.85–0.87
Prairie/savanna grass	0.80–0.95
Forest/woodland	0.90–1.00

amount of soil moisture which can be extracted by a plant cover when water is not limiting is defined by a root constant (C); typical values are given in Table 2.2 (Grindley, 1969). Actual evapotranspiration taking place as a soil dries out can be estimated using the model of Penman (1949) whereby actual evapotranspiration equals potential ($E_a = E_p$) as long as SMD < C but when SMD > C, a further 25 mm of moi. ture can be extracted at a reduced rate until, at SMD > $3C$, extraction becomes minimal ($E_a = 0.1\,E_p$).

Although the ability of vegetation to reduce soil moisture is recognized qualitatively, it is hard to quantify. Reduced soil moisture increases soil suction which affects both hydraulic conductivity and pore-water pressure. Only limited information is available, however, on differences in the hydraulic conductivity of soils with and without a vegetation cover and the effect of vegetation on slope stability through soil moisture depletion is difficult to separate from that of soil reinforcement by the rooting system. Nevertheless, through modification of the soil moisture content, vegetation affects t' frequency at which the soil becomes saturateu which, in turn, controls the likelihood of runoff generation or mass soil failure. The strength of this effect depends upon the local soil and climatic conditions and the vegetation type. It will also show, often substantial, seasonal variation, being greatest in summer and lowest in winter or whenever the vegetation is dormant.

2.2.2 INTERCEPTION

On contact with the canopy of a vegetation cover, the rainfall is divided into two parts. These are (1) direct throughfall, that which reaches the ground after passing through gaps in the canopy, and (2) interception, that which strikes the vegetation cover. If it is assumed t

Table 2.2 Values of the root constant (C) for use in estimating evapotranspiration (after Grindley, 1969)

Vegetation	*Maximum SMD (mm)*[a]	*Root constant, C (mm)*
Cereals	200	140
Temporary grass	100	56
Permanent grass	125	75
Rough grazing	50	13
Trees (mature stand)	125–250	75–200

[a] SMD = soil moisture deficit. The actual value of maximum SMD varies with the depth of roots, being higher for deep-rooted vegetation than for shallow-rooted types.

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the rain falls vertically, the volume of rainfall intercepted (*IC*) can be calculated from the simple relationship:

$$IC = \mathrm{RAIN} \cdot CC, \tag{2.1}$$

where CC = percentage canopy cover.

Some of the intercepted rainfall is stored on the leaves and stems and is later returned to the atmosphere by evaporation. The remainder of the intercepted rainfall, termed 'temporarily intercepted throughfall' (TIF), reaches the ground either as stemflow (i.e. that running down the stems, branches or trunks of the vegetation) or as leaf drainage.

Interception storage

Observed interception storage (IC_{store}) varies widely, depending upon the type of vegetation and the intensity of the rain, but, during a storm, it increases exponentially to a maximum value (IC_{max}) in a relationship similar to that proposed by Merriam (1973):

$$IC_{store} = IC_{max}(1 - \exp R_{cum}/IC_{max}), \tag{2.2}$$

where R_{cum} is the cumulative rainfall received since the start of the storm. Values of maximum interception storage are difficult to determine but probably range from 0.5 mm for deciduous forest in winter to 1 mm for coniferous forest, deciduous forest in summer and many agricultural crops, 1–2 mm for grasses and 2.5 mm for a multi-layered tropical rain forest (Table 2.3). Since storage often returns to the maximum value between storms, its cumulative effect over a year can be considerable and can account for 10–15% of the annual rainfall in cool-temperate hardwood forests, 15–25% in temperate broad-leaved forests, 20–25% for cereals and grass covers, and 25–30% in temperate coniferous and in tropical rain forests. Interception storage thus reduces the volume of rainfall reaching the ground surface by these amounts.

Stemflow

The amount of water shed by stemflow depends upon the angle of the stems of the plant to the ground surface (De Ploey, 1982; van Elewijck, 1989). For plants where the stem diameters are less than the median volume drop size of the rainfall, such as grasses, stemflow is at a maximum when the stem angles are between 50° and 70°. For plants with larger diameter stems, the situation is less clear. Van Elewijck (1988) recorded maximum stemflow on maize leaves at stem angles between 10° and 20° and on simulated branches at stem angles between 5° and 15° whereas Herwitz (1987) found that stemflow on branches (>4 cm diameter) of *Toona australis* and *Aleurites moluccana* increased linearly with stem angle to reach a maximum at a branch angle of 60°, the highest angle used in his experiments.

Very little information exists on volumes of stemflow. Measurements by Noble (1981) and Finney (1984) show stemflow volumes to be about 3–7% of storm rainfall for both Brussels sprouts with canopy covers of 40–50% and potatoes with 20–25% canopy. Higher values were observed for sugar beet at 42% of storm rainfall with 28% canopy cover (Finney, 1984). A figure of 55% was also recorded for sugar beet by Appelmans, van Hove and De Leenheer (1980). Values of 44% and 31% were recorded by Bui and Box (1992) in laboratory experiments under maize and sorghum respectively. High stemflow volumes can therefore be expected for plants with an architecture designed to concentrate water at their base and characterized by stems and leaves which converge towards the ground. De Ploey (1982) estimates that tussocky grasses may produce stemflow volumes that amount to 50–100% of the intercepted rainfall and Herwitz (1987) found that more than 80% of the impacting rain on tree branches inclined at 60° contributed to stemflow. Such concentrations of rainfall over relatively small areas can increase the effective rainfall intensity locally beneath tussocky grasses to 150–200% of that received at the top of the canopy (De Ploey, 1982). Even greater concentrations can occur in forests. Herwitz (1986) recorded an instance in the tropical rain forest in northern Queensland where stemflow fluxes measured during a rain-

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Table 2.3 Interception storage capacity for different vegetation types (after Horton, 1919; Leyton, Reynolds and Thompson, 1967; Zinke, 1967; Rutter and Morton, 1977; Herwitz, 1985)

Vegetation type	*Interception storage capacity, IC_{max} (mm)*
Fescue grass	1.2
Molinia	0.2
Rye grass	2.5
Meadow grass, clover	2.0
Blue stem grass	2.3
Heather	1.5
Bracken	1.3
Tropical rain forest	0.8–2.5
Temperate deciduous forest (summer)	1.0
Temperate deciduous forest (winter)	0.5
Needle leaf forest (pines)	1.0
Needle leaf forest (spruce, firs)	1.5
Evergreen hardwood forest	0.8
Soya beans	0.7
Potatoes	0.9
Cabbage	0.5
Brussels sprouts	1.0
Sugar beet	0.6
Millet	0.3
Spring wheat	1.8
Winter wheat	3.0
Barley, rye, oats	1.2
Maize	0.8
Tobacco	1.8
Alfalfa	2.8
Apple	0.5

fall of 11.8 mm in 6 min gave local depth equivalents of between 83 and 1888 mm. These large quantities of water beneath plants can play an important role in the generation of runoff.

Based on the work of van Elewijck (1988), the volume of stemflow (SF) may be estimated as a function of the average angle of the plant stems to the ground (PA) using the following equations:

for stem diameters < median volume drop diameter:

$$SF = TIF\ (\cos PA \cdot \sin^2 PA); \qquad (2.3)$$

for stem diameters > median volume drop diameter:

$$SF = TIF \cos PA. \qquad (2.4)$$

In the above, sin PA expresses the effect of gravity and cos PA expresses the effect of the projected length of the leaves and stems on the plant.

Leaf drainage

The volume of leaf drainage is equal to the volume of temporarily intercepted throughfall less the volume of stemflow. Leaf drainage comprises raindrops that are shattered into small droplets immediately they strike the vegetation and large drops formed by the temporary storage and coalescence of raindrops on the leaf

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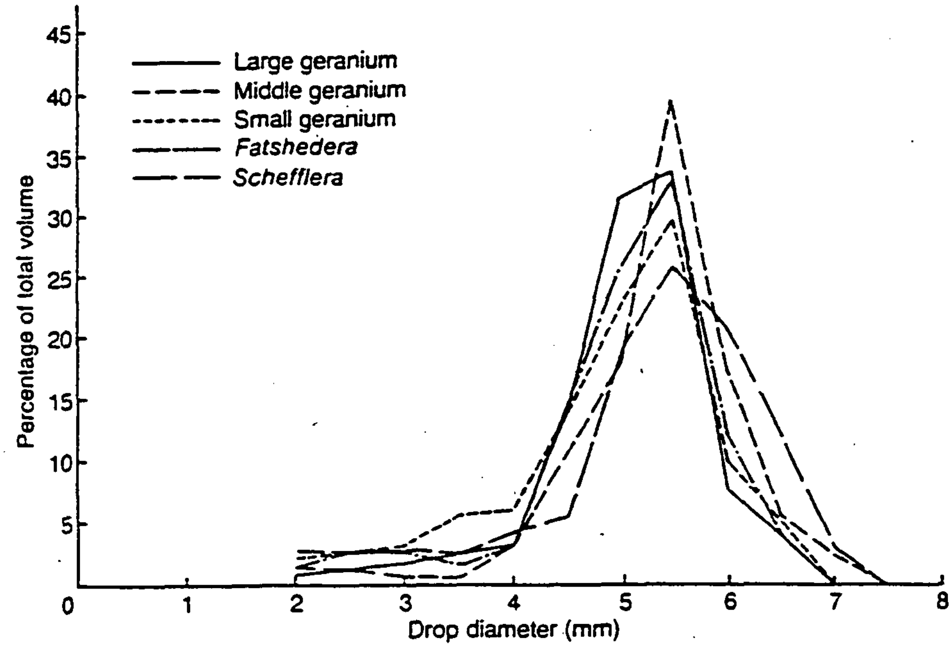

Figure 2.4 Drop-size distribution of leaf drainage (after Brandt, 1989).

and stem surfaces before they fall to the ground. Thus the rainfall beneath a plant canopy has higher proportions of small (<1 mm) and large (>5 mm) drops and fewer medium-sized drops compared with the original rainfall. In this way the canopy cover changes the drop-size distribution of the rain.

For plants with long leaves, like maize, the drops are mainly channelled along the centre vein and form leaf drips with diameters of 5–5.5 mm. For soya beans, the average size of the leaf drips is smaller, at about 4.5 mm, partly because more raindrops are rejected instantaneously by the leaves (Armstrong and Mitchell, 1987). Brandt (1989), in a review of previous literature combined with results of her own laboratory studies, concludes that leaf drainage has a normal drop-size distribution with a mean volume drop diameter of between 4.52 and 4.95 mm and a standard deviation of 0.79–1.30 mm (Figure 2.4).

Concentrations of water from leaf drip points can result in very high localized rainfall intensities, over 1000% greater than the intensity received at the canopy (Armstrong and Mitchell, 1987). These can exceed infiltration rates and result in surface runoff. This effect would be most marked in calm conditions. In strong winds, movement of the leaves and branches, as well as the falling water drops, will help to spread the leaf drainage more uniformly.

Soil detachment by raindrop impact

Soil detachment by raindrop impact has been related to various properties of the rain; *KE* (kinetic energy), EI_{30} (kinetic energy times the maximum intensity of the storm, measured over 30 min) and I^2 (intensity squared) being the most commonly used parameters. Vegetation affects these properties by altering the mass of rainfall reaching the ground, its drop-size distribution and its local intensity.

The energy of the rainfall available for soil detachment under a vegetation cover is dependent upon the relative proportions of the rain falling as direct throughfall and as leaf drainage. The ability of stemflow to detach soil particles is normally ignored. Thus the kinetic energy

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of the rain can be expressed by the simple arithmetical relationship:

$$KE = [(DT/TV) \cdot KE(DT)] + [(LD/TV) \cdot KE(LD)], \quad (2.5)$$

where KE = the kinetic energy (J/m^2 mm) of the rain; DT = the volume of direct throughfall; LD = the volume of leaf drainage; and TV = the total volume of direct throughfall and leaf drainage.

The energy of the direct throughfall is assumed to be the same as that of the natural rainfall. A reasonable approximation of the drop-size distribution of steady rain in temperate mid-latitude climates is that described by Marshall and Palmer (1948):

$$N(\delta) = N_0 e^{-\Lambda\delta}, \quad (2.6)$$

where $N(\delta)d\delta$ = the number of drops per unit volume with diameters between δ and $\delta + d\delta$; $\Lambda(I) = 41\,I^{-0.21}$, where Λ has units of cm^{-1} and I is the rainfall intensity (mm/h); and N_0 = approximately 0.08 cm^{-4}.

Other drop-size distributions have been presented by Carter *et al.* (1974) for Florida, Hudson (1963) for Zimbabwe, and Kowal and Kassam (1976) for northern Nigeria. In the case of the Marshall–Palmer distribution, the kinetic energy (J/m^2 mm) of a unit rain can be estimated from (Brandt, 1990):

$$KE(DT) = 8.95 + 8.44 \log I, \quad (2.7)$$

where I = the intensity of the rain (mm/h).

If the drop-size distribution of the leaf drainage follows that described above, its energy may be calculated from (Brandt, 1990):

$$KE(LD) = (15.8 \cdot PH^{0.5}) - 5.87, \quad (2.8)$$

where PH = the effective height (m) of the vegetation canopy.

For non-cohesive soils, the rainfall energy is not spent on detaching individual soil particles from the soil mass. It is primarily used for deformation of the surface and the lifting and moving of the already-discrete particles. In this case, splash erosion can be expected to be proportional to the kinetic energy of the rain (Free, 1960; Moss and Green, 1987), which is approximately proportional to $I^{1.14}$. Soil detachment (DET; g/m^2), in the sense of dislodgement of soil particles by raindrop impact, can then be estimated from the simple relationship:

$$\mathrm{DET} = k \cdot KE^{1.0} \cdot e^{-ah}, \quad (2.9)$$

where k = an index of the detachability of the soil (g/J); h = the depth (m) of the surface water layer, if any; and a = an experimental coefficient varying between 1.0 and 3.0 in value, depending upon the soil texture (Torri, Sfalanga and Del Sette, 1987).

It follows from this analysis that the ı. . of soil detachment beneath a vegetation cover depends upon the percentage canopy area, which controls the volumes of direct throughfall and leaf drainage, and the height of the canopy, which determines the energy of the leaf drainage. Numerous studies have shown that the energy of rain under vegetation can exceed that of an equivalent rainfall in open ground, both for trees (Chapman, 1948; Wiersum, Budirijanto and Rhomdoni, 1979; Maene and Chong, 1979; Mosley, 1982) and for lower-growing agricultural crops (Noble and Morgan, 1983; Morgan, 1985) with consequent increases in the rate of detachment (Finney, 1984; Wiersum, 1985). Field measurements with rainfall simulation showed that soil detachment under ır increased with percentage cover to double that recorded on bare soil when the canopy reached about 90% cover and was about 2 m above ground level (Morgan, 1985).

Recent research (Styczen and Høgh-Schmidt, 1988) has suggested that kinetic energy may not be the best parameter of the rain to explain soil detachment under vegetation or on cohesive soils. A different approach is proposed in which soil detachment is proportional to the sum of the squared momenta of the raindrops:

$$\mathrm{DET} = A(2\hat{e})^{-1} Pr \sum^{\delta} N_\delta p_\delta^2, \quad (2.10)$$

where A = a soil-dependent constant of proportionality; $\hat{e}$ = the average energy require-

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Table 2.4 Values of squared momentum for different intensities of rain

Rainfall intensity, I (mm/h)	*Squared momentum, M_R $((N\,s)^2/m^2\,s)$*
5	2.66×10^{-7}
10	8.88×10^{-7}
20	2.86×10^{-6}
35	7.11×10^{-6}
50	1.25×10^{-5}
75	2.32×10^{-5}
100	3.56×10^{-5}
125	4.92×10^{-5}
150	6.38×10^{-5}
175	7.93×10^{-5}
200	9.55×10^{-5}
225	1.12×10^{-4}
250	1.30×10^{-4}

break the bonds between two micro-aggregates of soil, and the energy lost by heat in the process; Pr = the probability that the kinetic energy received by the detached micro-aggregate(s) is large enough to make it measurable as splash, i.e. to make the micro-aggregate jump a minimum distance; N_δ = the number of raindrops of size (diameter) δ; and p_δ = the drop momentum ($m_\delta \cdot v_\delta$).

A, $\hat{e}$ and Pr are related to soil properties, while N_δ and p_δ are rainfall properties; m and v refer respectively to the mass and velocity of the raindrop.

For the Marshall–Palmer drop-size distribution, $N_\delta p_\delta^2$ is proportional to $I^{1.63}$ for $0 < I < 100$ mm/h and $I^{1.43}$ for $100 < I < 250$ mm/h. Values for the squared momentum, $M_R = N_\delta p_\delta^2$, are listed in Table 2.4.

The squared momentum of the leaf drainage (M_{RC}) can be calculated in the following way (Styczen and Høgh-Schmidt, 1986), given that the amount of leaf drainage equals $CC \cdot I \cdot [1 - (SF + IC_{store})]$ and the number of drops equals $CC \cdot I \cdot [1 - (SF + IC_{store})/\mathrm{vol}(\delta)]$ (equation 2.11) where vol(δ) = the volume of a drop with diameter (δ); ρ_w = the density of water; $v_{\delta H}$ = the velocity of the drop as a function of its diameter (δ) and fall height (H); and vol δ) $\rho_w^2 v_{\delta H}^2 = DH$, listed in Table 2.5.

$$M_{RC} = \frac{CC \cdot I \cdot [1 - (SF + IC_{store})]}{\mathrm{vol}(\delta)} \cdot \rho_w^2 \quad [\mathrm{vol}(\delta)^2 \cdot v_{\delta H}^2]. \qquad (2.11)$$

When the sum of the squared momenta with and without a vegetation cover are known, the

Table 2.5 Values of the parameter DH $(\rho_a^2 \pi \delta^3 v_\delta^2/6)(\mathrm{kg}^2/\mathrm{m\,s})$ computed for different drop sizes (δ) and fall heights

Fall height (m)	*Drop sizes (δ)*			
	4.5 mm	*5.0 mm*	*5.5 mm*	*6.0 mm*
0.5	0.4180	0.5734	0.7633	0.9909
1.0	0.7942	1.1002	1.4787	1.9384
1.5	1.2120	1.6890	2.2836	2.9996
2.0	1.5720	2.1866	2.9508	3.8837
3.0	2.1291	2.9998	4.0757	5.4158
4.0	2.5706	3.6229	4.9526	6.6014
5.0	2.9029	4.1470	5.6452	7.4386
6.0	3.1459	4.4763	6.0883	8.0182
7.0	3.2949	4.6733	6.3533	8.4036
8.0	3.3907	4.7957	6.5331	8.6590
9.0	3.4554	4.8971	6.6696	8.8381
10.0	3.5125	4.9769	6.7768	8.9584
13.0	3.6530	5.1843	6.9936	9.2016
∞	3.8647	5.4080	7.2934	9.5310

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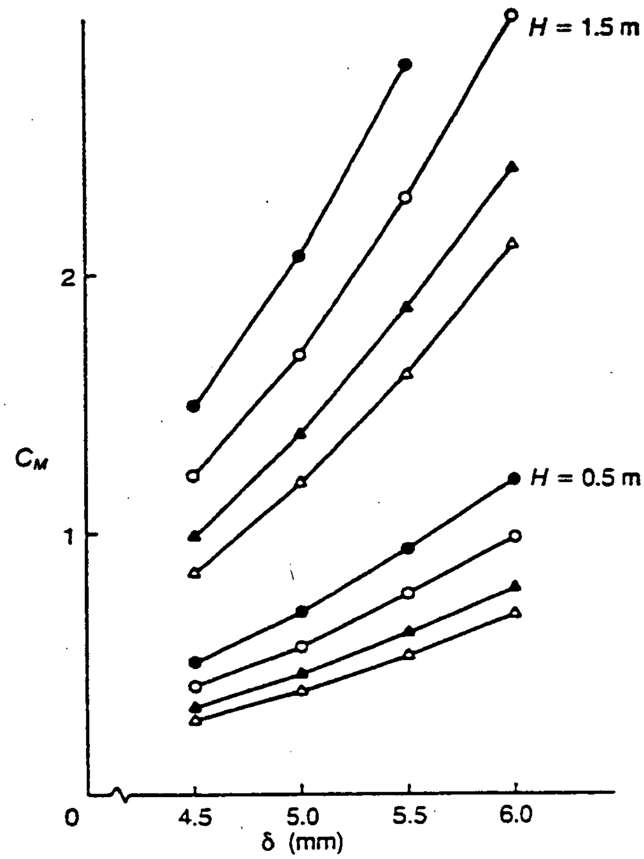

Figure 2.5 C_M as a function of the drop size of transformed rain (δ) for different rainfall intensities (I) and two canopy heights (H) (after Styczen and Høgh-Schmidt, 1988). The canopy cover is 100%. Storage and stemflow are estimated as 10% of the rainfall. ●, I = 35 mm/h; ○, I = 50 mm/h; ▲, I = 75 mm/h; △, I = 100 mm/h.

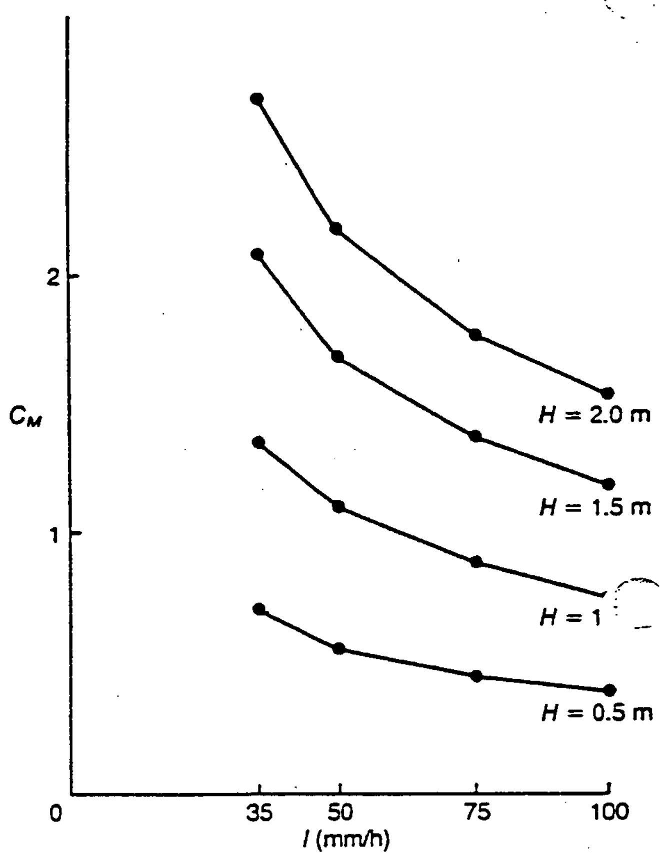

Figure 2.6 Changes in C_M with changes in rainfall intensity (I) for different canopy heights (H) (after Styczen and Høgh-Schmidt, 1988). Curves are calculated for 100% canopy cover and a drop size for leaf drainage (δ) of 5.0 mm.

relative effect of the vegetation on splash (equivalent to a C-factor for splash) can be calculated as:

$$C_M = \frac{(1-CC)M_R + C \cdot I \cdot (1-(SF+IC_{store})) \cdot DH}{M_R}. \tag{2.12}$$

Figures 2.5, 2.6 and 2.7 illustrate the calculated effect of different drop sizes, fall heights, stemflow percentages and rainfall intensities on the value of C_M. Figure 2.8 shows how important the drop-size distribution of the rain can be when interpreting the effects of vegetation. As splash erosion is proportional to the drop diameter raised to the sixth power, leaf drainage may result in serious soil breakdown. Contrary to ordinary opinion, a plant canopy situated more than about 1 m above the ground cannot be expected to decrease splash erosion by its indeed, it is more likely to enhance it. Figure 2.9 shows this effect measured under maize (Morgan, 1985) and calculated according to equations 2.10 and 2.12.

Similar conclusions were reached by Moss and Green (1987) who, on the basis of empirical data, divided vegetation layers into the following categories:

1. Layer 1, <0.3 m: where, owing to the often high density of plant-ground contacts, leaf drainage volumes are usually small and the impact velocities too low to allow significant damage.

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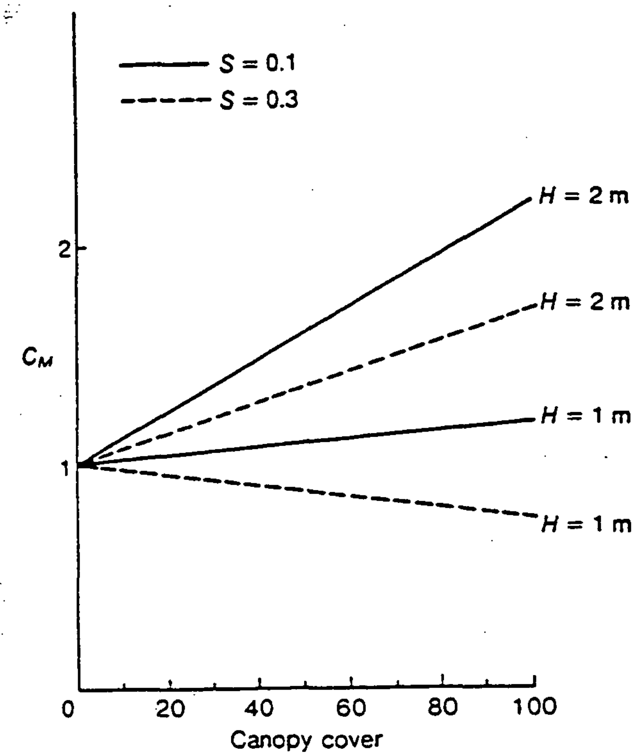

Figure 2.7 Changes in C_M with changing percentage permanent interception (S) for leaf drainage with a diameter (δ) of 5.0 mm and two heights of canopy (H) (after Styczen and Høgh-Schmidt, 1988). Rainfall intensity equals 50 mm/h.

2. Layer 2, 0.3–1.0 m: where there is a transition from small to significant leaf drip and soil damage.
3. Layer 3, 1.0–2.5 m: from which leaf drips reach high erosivity and achieve a marked ability to cause soil damage.
4. Layer 4, 2.5–6.0 m: in which the ability of leaf drips to cause erosion and soil damage continues to increase but more slowly than in layer 3.
5. Layer 5, >6 m: where the free fall height is sufficient for leaf drips to attain 90% or more of their terminal velocity; hence, above this height there is little further increase in either their ability to cause soil damage or their erosivity.

If, in contrast to the above, it is assumed that splash erosion on sand is proportional to the incoming kinetic energy instead of the sum of the squared momenta, the apparent effects of vegetation become less drastic. For very tall vegetation, the energy impact approximately doubles compared to that on bare soil, but for most agricultural crops, the impact is reduced. The relative change in energy impact is shown in Figure 2.10 for four rainfall intensities, five fall heights, 10% stemflow and different cover percentages. From such calculations, a soil under a 0.5 m tall soya bean crop (80% cover, 10% stemflow) receives only 50% of the energy received by a bare soil. In the case of 1.5 m tall maize (also 80% cover and 10% stemflow), the soil receives 85–90% of the energy. For 6 m tall trees without any ground cover, the energy received by the soil reaches 200%.

It may seem strange that the amount of energy reaching the ground under trees is more than 100%. This is due to the difference in frictional resistance on small and large drops. Leaf drips are not only larger and heavier, they also gain a higher velocity so that the final impact energy is increased.

Equation 2.10 contains two soil factors that may be influenced by vegetation. These are $\hat{e}$, the average energy required to break the bonds between two micro-aggregates, and Pr, the probability that the kinetic energy received by the micro-aggregate is large enough to make it measurable as splash. The term, k, in equation 2.9, which expresses the detachability of the soil, also encompasses these factors which are discussed in more detail in section 2.4.1.

2.2.3 INFILTRATION

For serious erosion to take place, some amount of runoff must occur. The amount of runoff generated is closely related to the infiltration rates (unsaturated and saturated hydraulic conductivity) of the soil, the antecedent moisture content and, indirectly, to the direction of water flow within the soil.

When rain water reaches the ground underneath vegetation, it may stand a better chance of infiltrating than on unvegetated soil. Organic

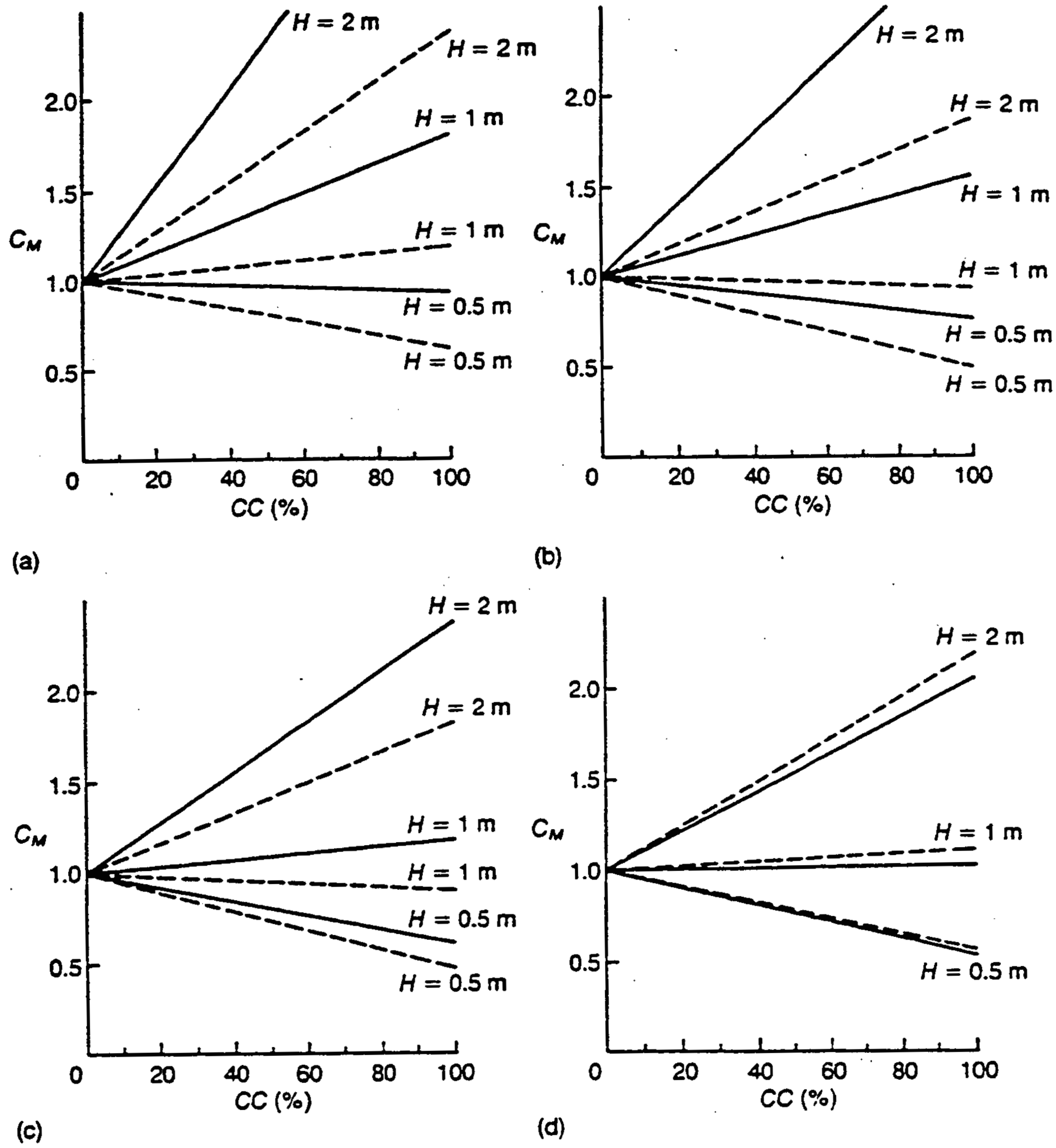

Figure 2.8 C_M as a function of percentage vegetation cover (*CC*) and canopy height (*H*), calculated for two drop-size-distributions (—, Marshall and Palmer, 1948; ---, Carter *et al.*, 1974) for leaf drainage of 5.0 mm diameter (δ), and percentage permanent interception storage and stemflow equal to 10% of the rainfall (after Styczen and Høgh-Schmidt, 1988). Rainfall intensities are (a) 35 mm/h; (b) 50 mm/h; (c) 75 mm/h; (d) 100 mm/h.

matter, root growth, decaying roots, earthworms, termites and a high level of biological activity in the soil help to maintain a continuous pore system and thereby a higher hydraulic conductivity. Through an increase in the infiltration rate, and perhaps also in the moisture storage capacity of the soil, vegetation may decrease the amount of runoff generated during a storm; it will probably also increase the time taken for runoff to occur. A bare soil may be compared to a bucket with few or small holes in the bottom, while the vegetated soil is rather like a slightly larger bucket with more and bigger holes. It is necessary to apply more water at a greater rate to make the second bucket overflow. Thus, a higher infiltration may decrease

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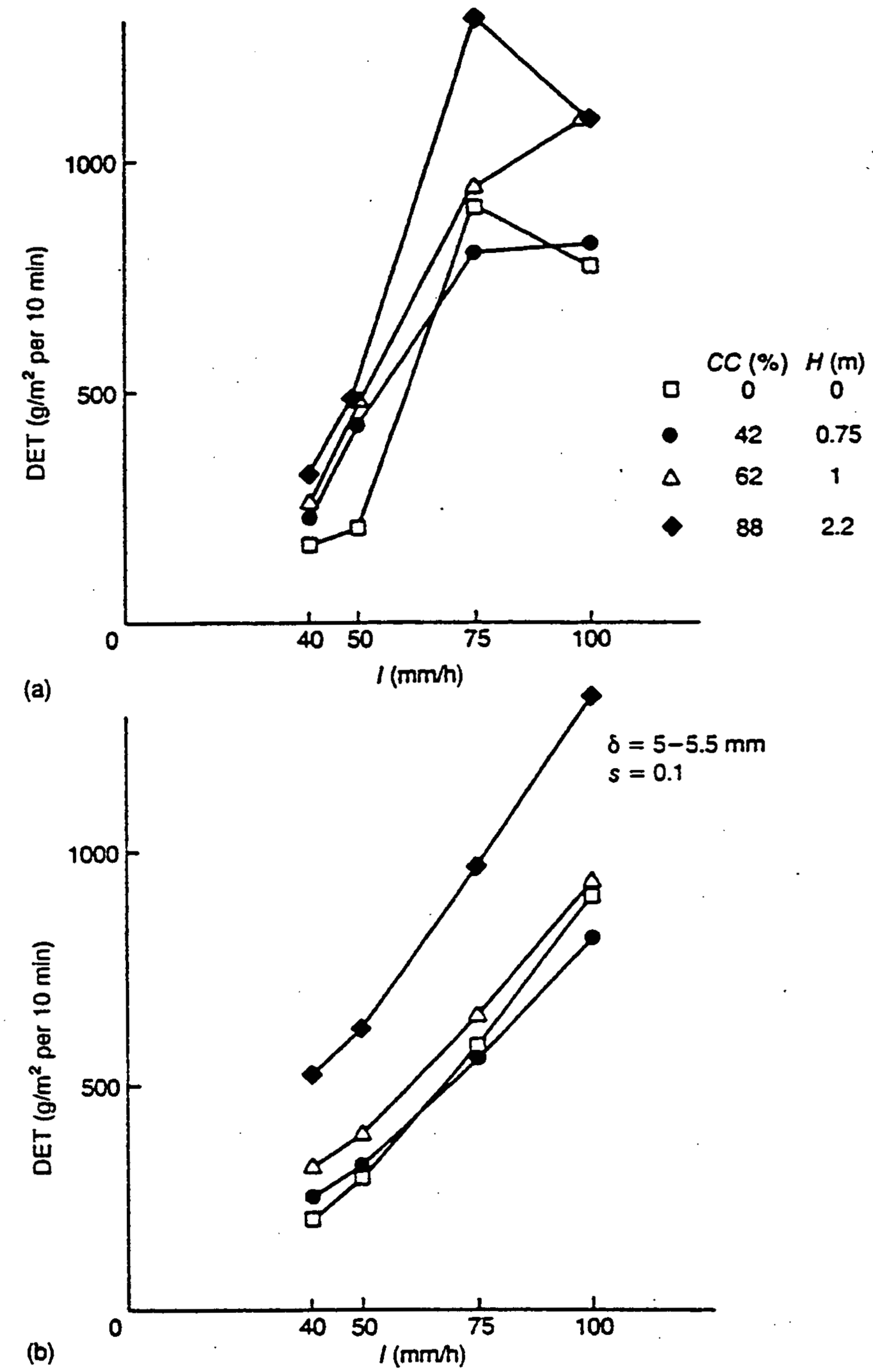

Figure 2.9 Splash erosion (DET) as a function of rainfall intensity for four combinations of percentage maize cover (CC%) and height (*H*) (after Styczen and Høgh-Schmidt, 1988). (a) Observed data (from Morgan, 1985); (b) calculated data. Based on equations 2.10 and 2.12 with a drop diameter (δ) of leaf drainage of 5.0–5.5 mm and percentage permanent interception storage and stemflow equal to 10% of the rainfall.

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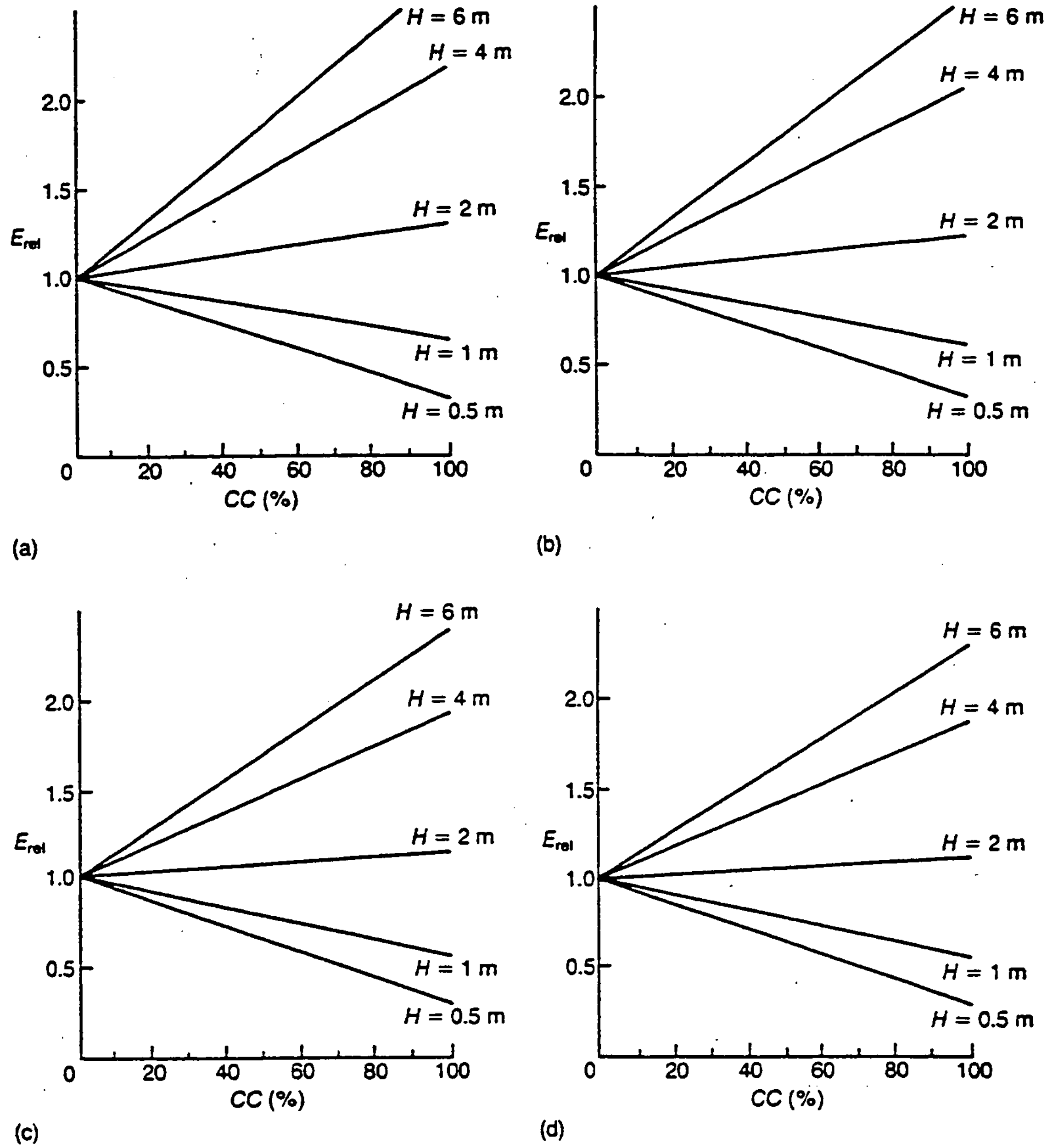

Figure 2.10 Relative change in kinetic energy of the rain (E_{rel}) reaching the soil as a function of percentage vegetation cover (CC%) and canopy height (H), calculated for the Marshall–Palmer drop-size distribution, drop diameter (δ) of leaf drainage of 5.0 mm and percentage permanent interception and stemflow of 10% of the rainfall. Rainfall intensities are (a) 35 mm/h; (b) 50 mm/h; (c) 75 mm/h; (d) 100 mm/h.

the number of erosive events per year because a greater storm is needed to produce the critical amount of runoff.

The saturated hydraulic conductivity of a soil (k_{sat}) depends on its texture and structure, the presence of cracks and the number of biopores it contains. McKeague, Wang and Coen (1986) present some guidelines for estimating k_{sat} from soil morphology. These are of interest here because the descriptions help in visualizing the changes occurring in a soil as a result of biological interference. Rawls, Brakensiek and Soni (1983) and Brakensiek and Rawls (1983) estimate k_{sat} for soils with different pore-size distri-

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Table 2.6 Morphological descriptions and corresponding values of saturated hydraulic conductivity (k_{sat}) for soils with a loamy texture but different structural properties and pore content (from McKeague, Wang and Coen, 1986)

k_{sat} *(mm/h)*	*Soil description*
1.5–5.0	Massive to weak coarse blocky or prismatic non-compact loamy or clayey material with tightly-accommodated peds (if any), < 0.02% channels > 0.5 mm, and few very fine voids visible with a hand lens
5.0–15	Structureless loamy material, friable, bulk density $1.5 \pm 0.1\,Mg/m^3$, not compact, with < 0.02% channels
15–50	*Either* moderately packed loamy to clayey material with weakly developed pedality (adherent partly-formed peds); 0.02–0.1% channels ⩾ 0.5 mm, some of which traverse the horizon
	or moderate medium to coarse blocky loamy or clayey material with firm dense peds, < 0.02% channels (= to 15 biopores/m^2 of 4 mm diameter)
50–150	*Either* approximately 0.1–0.2% channels > 0.5 mm, at least half of which extend through the horizon, < 0.02% large (⩾ 5 mm) channels, structureless or weak structure; texture finer than fine sandy loam, if not compact
	or moderate fine or medium blocky with weakly adherent peds or moderate to strong medium to coarse blocky; < 0.1% channels extend through the horizon; texture finer than fine sandy loam, if not compact

butions, textures and organic matter contents, with and without tillage, and with crusting.

In Table 2.6, six different morphological descriptions of loamy soil are given, together with corresponding values for saturated hydraulic conductivity (McKeague, Wang and Coen, 1986). The soils have the same texture but differ with respect to the number of biopores or the level of aggregation and structural development. According to this description, the saturated hydraulic conductivity of a loamy soil may range from 1.5 to 150 mm/h, depending on these soil characteristics.

It is evident that such large differences in hydraulic conductivity will result in large differences in the amount of runoff generated during a particular storm. Assuming that the soil is already wet and the rate of infiltration equals the saturated hydraulic conductivity of the soil, a very simple way to estimate the amount of runoff is to compare rainfall intensity and infiltration ($I - k_{sat}$). Runoff amounts for different intensities and four values of k_{sat} are given in Table 2.7. Similar results can be calculated for soils of other textures.

For dry soils, the differences in runoff generation may be even larger, as the time taken to wet the soil to saturation varies. This also implies that the delay in time before runoff occurs is longer for soils with high hydraulic conductivities.

At present it is possible to account for the effects of vegetation on infiltration only very crudely, for example by an empirical adjustment to the value of k_{sat} for an unvegetated soil, so that (Holtan, 1961):

$$k_{satveg} = k_{sat} + a^{1.4}, \qquad (2.13)$$

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Table 2.7 Runoff amounts (m^3/ha h) calculated for eight rainfall intensities and four infiltration rates

	Infiltration rate (mm/h)			
Rainfall intensity (mm/h)	*1.5*	*5.0*	*15*	*50*
10	85	50	0	0
20	185	150	50	0
30	285	250	150	0
40	385	350	250	0
50	485	450	350	0
75	735	700	600	250
100	985	950	850	500
150	1485	1450	1350	1000

Table 2.8 Basal areas for different vegetation types (after Holtan, 1961)

Land use or cover	*Hydrological condition*	*Percentage basal area rating*
Fallow (after row crop)	-	0.10
Fallow (after sod)	-	0.30
Row crops	poor	0.10
	good	0.20
Small grain	poor	0.20
	good	0.30
Hay (legumes)	poor	0.20
	good	0.40
Hay (sod)	poor	0.40
	good	0.60
	excellent	0.80
Pasture or range (bunch grass)	poor	0.20
	fair	0.30
	good	0.40
Temporary pasture (sod)	poor	0.40
	fair	0.50
	good	0.60
Permanent pasture or meadow (sod)	poor	0.80
	good	1.00
Woods and forest	-	1.00

where k_{satveg} = the saturated hydraulic conductivity of the soil with a vegetation cover; and a = percentage basal area of the vegetation.

Typical values of a are given in Table 2.8. This adjustment alters k_{sat} largely as a function of vegetation density and, broadly, lumps together all the effects of vegetation mentioned above that lead to higher infiltration rates and, probably, though it is by no means clear, to a reduction in the amount of rain reaching the ground after interception by the vegetation canopy. It does not take account of the effect of the vegetation cover on the spatial distribution of the rainfall at the ground surface which, through concentrations of water in leaf drips and stemflow, can lead, as seen above, to localized intensities which may exceed infiltration capacity and result in runoff generation. Alter-

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natively, k_{satveg} could be measured in the field but it is often difficult to place infiltrometers over representative areas of the vegetation cover, particularly with shrubs and scrub. Also, the variability in k_{satveg} is invariably great, with coefficients of variation in excess of 200%, so that large numbers of replicates are required to obtain meaningful values.

A further way in which vegetation influences infiltration is through the difference in antecedent moisture content that may occur because more water is removed by evapotranspiration from a soil covered by vegetation than from a bare soil surface. Thus, the capillary pressure, ψ, within the soil at the onset of rain may be lower (ψ numerically higher) and the time before saturation is reached longer for a vegetated soil.

2.2.4 SURFACE CRUSTING

On silty soils, soils containing high proportions of fine sand, soils low in organic matter and soils which for some other reason have an unstable or poor structure, surface crusting or sealing may take place, as the finer particles detached by raindrop impact clog up the pores and cracks and reduce the infiltration rate. The speed of crusting seems to be related to either the incoming shear forces of the raindrops (Farres, 1978; Al-Durrah and Bradford, 1982) or the rainfall energy (Boiffin, 1985; Govers and Poesen, 1985).

The effect of raindrop impact on infiltration can be particularly spectacular on soils susceptible to crusting. Brakensiek and Rawls (1983) present data on soils with and without crusts, where the crusted soils sustain infiltration rates 15–20 mm/h lower than the uncrusted soils. Measurements on sandy soils in Israel show that crusting reduces the infiltration capacity from 100 to 8 mm/h, and on a loess soil from 45 to 5 mm/h (Morin, Benyamini and Michaeli, 1981). The infiltration capacity of sandy soils in Mali ranges from 100 to 200 mm/h but, when a crust has developed, it is reduced to 10 mm/h. Only a few storms are needed to bring about this change. A 50% reduction in infiltration can occur in one storm (Hoogmoed and Stroosnijder, 1984). Studies on the behaviour of loamy soils in northern France (Boiffin, 1985) show that surface crusting can reduce infiltration capacities from 20–50 mm/h to about 1 mm/h at a rate which is dependent upon the cumulative rainfall received since tillage.

Considering the effect of vegetation on the rainfall energy (Figure 2.10), it may be expected that the speed at which the infiltration rate is reduced is altered by a plant cover. Under relatively low vegetation, the infiltration may remain higher for a longer time than on bare soil, resulting in runoff occurring later and in smaller quantities. Under tall vegetation without undergrowth, the opposite situation may prevail, because sealing or crusting will take place shortly after the onset of rain.

2.3 HYDRAULIC EFFECTS

The passage of water across a bare soil surface may entrain and transport soil particles already detached and, particularly if the flow is concentrated in channels, may also detach additional particles. Erosion is said to be either detachment-limited or transport-limited, depending on how much detached material is available for transport at a given moment. If too much material exists, all of it will not be removed, the erosion rate will be controlled by the transport capacity and the erosion will be transport-limited. If the transport capacity exceeds the detachment rate, all the detached particles will be removed, the erosion rate will be controlled by the supply of detached material and will be detachment-limited.

Flow transport capacity and detachment by flow are often easily confused. Both are related to energy-spending processes within the flow at the interface between the flowing water and the bed. There are differences, however, in the way the energy is expended. Transport capacity is defined as the capacity of flow to carry material of a given noncohesive type (primary particles and individual soil aggregates) with the energy of the flow spent on lifting and carrying the

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sediment particles. Soil detachment by flow relates to the situation where the energy of the flow is spent on detachment as well as entrainment and transport of sediment particles. Vegetation can limit the capacity of flowing water to detach and transport sediment. The most obvious effect is through the reduction in flow velocity brought about by contact between the flow and the vegetation. The stems and leaves of the vegetation impart roughness to the flow.

2.3.1 SURFACE ROUGHNESS AND FLOW VELOCITY

Surface roughness is an important parameter controlling the speed of the generated runoff. It may be described by a coefficient of friction. The coefficient of friction is usually an 'effective' roughness coefficient that includes the effects of raindrop impact, concentration of the flow, obstacles such as litter, ridges, rocks and roughness from tillage, the frictional drag over the surface, and the erosion and transport of sediment (Engman, 1986). It is more a function of vegetation (plant arrangement, plant population, litter, mulch) and, on agricultural land, tillage methods, than it is a soil–vegetation interaction parameter. The roughness coefficient is normally considered as a summation of the roughness imparted by the soil particles, surface micro-topography (form roughness) and vegetation, acting independently of each other.

Surface roughness is inversely related to both the velocity and quantity of runoff as expressed by the following equations:

$$u = R^{0.667} S^{0.5} / n \tag{2.14}$$

and

$$Q = R^{1.667} S^{0.5} / n \tag{2.15}$$

where u = the velocity of the flow (m/s); R = the hydraulic radius (m), often taken as equal to flow depth in shallow flows; S = slope of the energy line (m/m); n = Manning's roughness coefficient ($m^{1/6}$); and Q = the quantity of runoff (m^3/m s).

Rewriting the equations shows that velocity is dependent on roughness to the power of -0.6 ($u \propto n^{-0.6}$).

For a given amount of runoff it may be calculated that doubling the roughness increases the water depth by 50% and decreases velocity by 34%.

Alternative friction factors to Manning's n are the dimensionless Darcy-Weisbach (f) and Chezy's (C). These are related to Manning's n and to each other as follows:

$$f = 8 g n^2 R^{-0.33} \tag{2.16}$$

$$C = R^{0.167} / n \tag{2.17}$$

and

$$C = (8g/f)^{0.5}. \tag{2.18}$$

Engman (1986) has suggested a number of values for Manning's roughness coefficient. These are listed in Table 2.9. The possible range of n is large: for bare smooth soil, n is in the order of 0.01; for 5–10 t/ha of straw mulch, $n = 0.07$; and for grass, n ranges from 0.2 to 0.4. Thus, for a constant amount of runoff, surface roughness reduces flow velocity on a mulched field to approximately one-third and on a grass field to one-eighth of what it would be on bare smooth soil.

The level of roughness depends upon the morphology of the plant and its density of growth. Manning's n values can be related to a vegetation retardance index (CI) which is a function of the density and height of the plant ste (Temple, 1982; Table 2.10). The range in n values for each retardance class reflects the variation in roughness which occurs with flow depth (Figures 2.11 and 2.12). With shallow flows, the vegetation stands relatively rigid and roughness values are about 0.25–0.3, associated with distortion of the flow around the individual plant stems. As flow depth increases, the stems begin to oscillate, further disturbing the flow, and roughness values rise to around 0.4. When flow depth begins to submerge the vegetation, roughness values decline rapidly, often by an order of magnitude, because the plants tend to lay down in the flow and roughness is mainly

Table 2.9 Recommended values for Manning's n (after Engman, 1986)

Cover or treatment	*Residue rate (t/ha)*	*Value recommended*	*Range*
Concrete or asphalt		0.011	0.01–0.013
Bare sand		0.01	0.010–0.016
Gravelled surface		0.02	0.012–0.03
Bare clay-loam (eroded)		0.02	0.012–0.033
Fallow – no residue		0.05	0.006–0.16
Chisel plough	<0.6	0.07	0.006–0.17
	0.6–2.5	0.18	0.07–0.34
	2.5–7.5	0.30	0.19–0.47
	>7.5	0.40	0.34–0.46
Disc/harrow	<0.6	0.08	0.008–0.41
	0.6–2.5	0.16	0.10–0.25
	2.5–7.5	0.25	0.14–0.53
	>7.5	0.30	-
No tillage	<0.6	0.04	0.03–0.07
	0.6–2.5	0.07	0.01–0.13
	2.5–7.5	0.30	0.16–0.47
Mouldboard plough		0.06	0.02–0.10
Coulter		0.10	0.05–0.13
Range (natural)		0.13	0.01–0.32
Range (clipped)		0.10	0.02–0.24
Grass (blue grass sod)		0.45	0.39–0.63
Short grass prairie		0.15	0.10–0.20
Dense grass		0.24	0.17–0.30
Bermuda grass		0.41	0.30–0.48

due to skin resistance; as a result, velocities increase.

Greatest reductions in flow velocity occur ith dense, spatially uniform vegetation covers. Very open, clumpy and tussocky vegetation is less effective and may even lead to localized increases in velocity and erosion as flow becomes concentrated between the clumps. Figure 2.13 shows typical flow paths around a clump of vegetation. As flow separates around the clump, pressure (normal stress) is higher on the upstream than on the downstream face, and eddying and turbulence are set up downstream; in addition, a zone of backflow is established just upstream of the clump (Babaji, 1987). Vortex erosion can occur both upstream and downstream. The combined effect of these processes means that erosion rates under tussocky vegetation may remain high and match those prevailing on bare soil (De Ploey, Savat and Moeyersons, 1976).

2.3.2 SEDIMENT TRANSPORT CAPACITY

A number of equations have been proposed for calculating the transport capacity of flow. Traditionally, transport capacity has been defined with respect to sandy beds (river beds) and some confusion arises when dealing with cohesive beds, partly because energy is also spent in detaching the material and partly because the mechanisms of detachment are less well understood than is the case with sand particles.

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Table 2.10 Values of Manning's *n* for different vegetation retardance classes (after Temple, 1982)

Vegetation type	*CI*[a]	*n*
Very tall dense grass (>600 mm)	10.0	0.06–0.20
Tall grass (250–600 mm)	7.6	0.04–0.15
Medium grass (150–250 mm)	5.6	0.03–0.08
Short grass (50–150 mm)	4.4	0.03–0.06
Very short grass (< 50 mm)	2.9	0.02–0.04

[a] *CI* = index of vegetation retardance defined by Temple, (1982):

$$CI = 2.5(h\sqrt{M})^{1/3}$$

where h is the height of the plant stems (m) and M is the density of stems (stems/m^2).

Reference stem densities for good uniform stands are:

Bermuda grass	5380
Buffalo grass	4300
Kentucky bluegrass	3770
Weeping love grass	3770
Alfalfa	5380
Common lespedeza	1610
Sudan grass	538

For legumes and large-stemmed or woody species, the reference stem density is about five times the actual count of stems very close to the bed.

The situation is complicated further because at least three main modes of transport may occur in the flow. Material may be transported as bed load, suspended load or wash load. Bed load is transported in a rolling manner along the soil surface. This mode of transport dominates when the ratio between the lifting forces and the stabilizing forces on the particle is below 0.2; in other words, when the particles are large and heavy compared to the forces within the flow. When this ratio is greater than 1.0, particles are transported as suspended load (Engelund and Hansen, 1967). For very small particles (less than approximately 2 μm), Brownian movements dominate over sedimentation following Stokes' Law, and this material is referred to as wash load.

A very large number of sediment transport equations have been published. Some were derived for sandy river beds, others for soils, but none covers all three modes of transport. The following are typical of those in which sediment transport is related to runoff.

Schoklitsch (1950): for bed load in rivers,

$$Qs = Cs S^{1.4} \gamma_w^2 Q^{0.6} (Q^{0.6} - Qc^{0.6}) n^{-0.5}, \tag{2.19}$$

where Cs = a coefficient expressing soil characteristics: γ_w = unit weight of water; and Q_c is

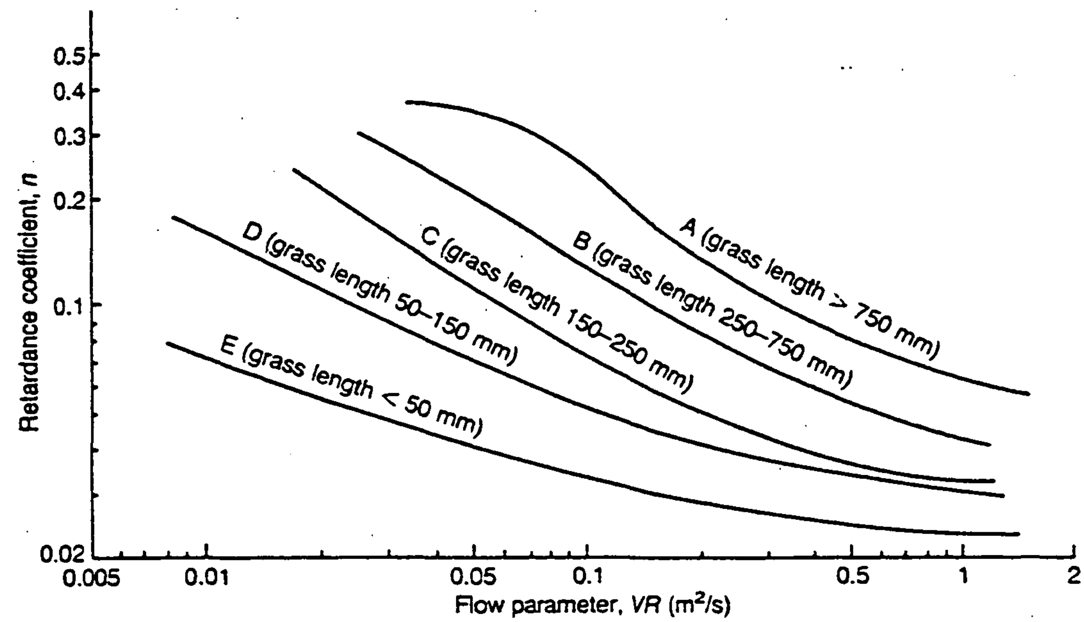

Figure 2.11 Variation in the frictional resistance of grass swards expressed by Manning's *n* for different retardance categories (after US Soil Conservation Service, 1954).

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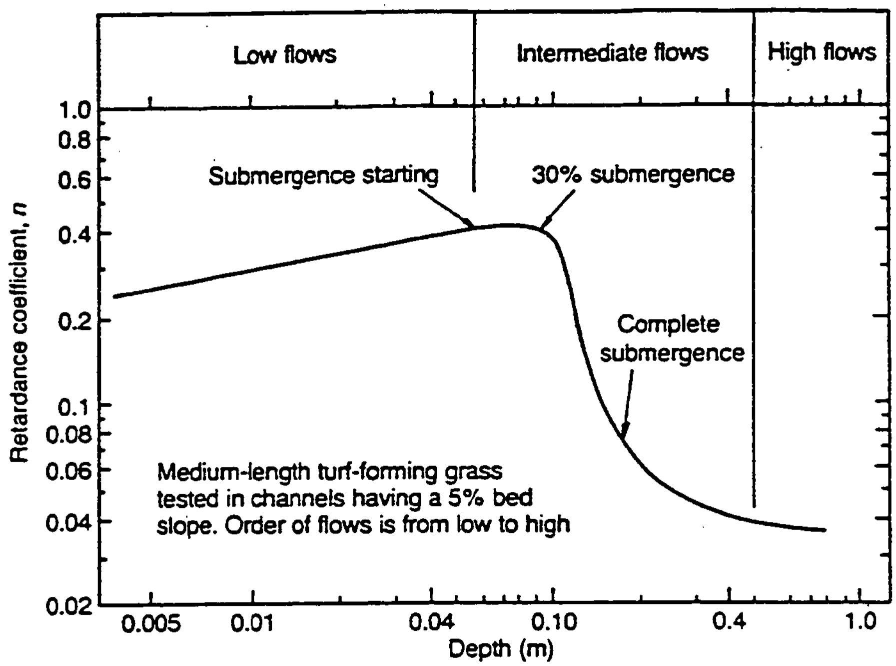

Figure 2.12 Relationship between Manning's n and depth of water flow for a medium length grass (after Ree, 1949).

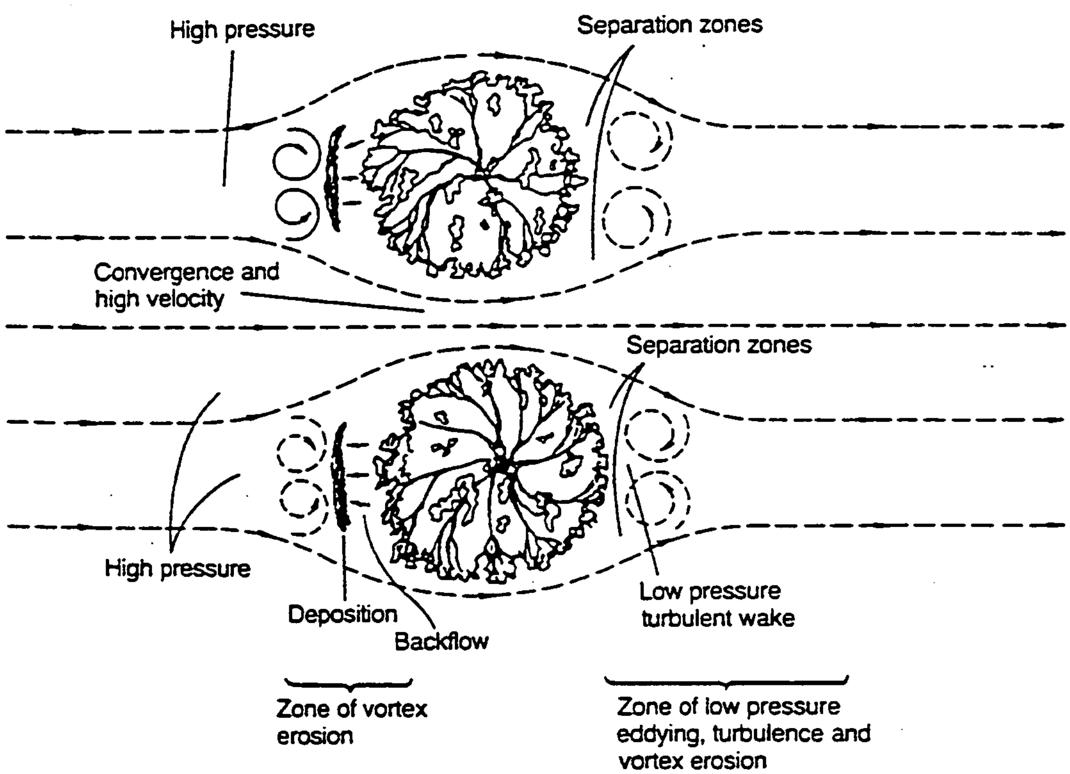

Figure 2.13 Plan view of the pattern of water flow around tussocky vegetation (after Babaji, 1987).

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Table 2.11 Relative transport capacities ($m^3/m\,s$) for runoff generated on loamy soil with different hydraulic properties. The runoff amounts used are given in Table 2.7

	Saturated hydraulic conductivity (mm/h)			
Precipitation (mm/h)	*1.5*	*5.0*	*15*	*50*
10	0.055	0.023	0	0
20	0.20	0.14	0.023	0
30	0.41	0.33	0.14	0
40	0.68	0.58	0.33	0
50	1.0	0.88	0.58	0
75	2.0	1.8	1.4	0.33
100	3.3	3.1	2.5	1.1
150	6.5	6.2	5.5	3.3

the critical discharge at which sediment transport takes place.

Meyer and Wischmeier (1969): for shallow flows on hillslopes,

$$Qs = kS^{5/3}Q^{5/3}, \qquad (2.20)$$

where k = an experimental coefficient largely related to soil characteristics.

Engelund and Hansen (1967): for bed load transport in rivers but applied by Nielsen and Styczen (1986) to runoff on hillslopes,

$$Qs = 0.04(2g/0.25f)^{1/6}(QS)^{5/3}/(J-1)^2 g^{1/2} d_{50}, \qquad (2.21)$$

where f = Darcy-Weisbach roughness coefficient; J = relative sediment density, defined as the ratio γ_s/γ_w, where γ_s is the volume weight of transported sediment; and d_{50} = the diameter at which 50% of the soil particles are finer.

De Ploey (1984): for overland flow and rill flow on hillslopes,

$$Qs/Q \propto S^{1.25}Q^{0.625}. \qquad (2.22)$$

Govers and Rauws (1986): for overland flow,

$$Qs/Q = A\,Su + B, \qquad (2.23)$$

where Su = unit stream power (product of slope and mean flow velocity); and A, B = empirical coefficients which vary in value with sediment particle size.

Carson and Kirkby (1972): for overland flow

$$Qs = 0.0085\,Q^{1.75}S^{1.625}d_{84}^{-1.11} \qquad (2.24)$$

where d_{84} = the diameter at which 84% of the soil particles are finer.

Morgan (1980): for overland flow,

$$Qs = 0.0061\,Q^{1.8}S^{1.13}n^{-0.15}d_{35}^{-1}, \qquad (2.25)$$

where d_{35} = the diameter at which 35% of the soil particles are finer.

In these equations, Qs = sediment transport capacity ($m^3/m\,s$) and Q = volume of runoff ($m^3/m\,s$), so that Qs/Q = sediment concentration.

Despite their differences and shortcomings, the equations consider transport capacity to be proportional to the volume of runoff raised to the power of between 1.6 and 1.8, or to $Q(Su - B)$. Transport capacity is also inversely related to both roughness, n, raised by powers of between 0.15 and 0.5, and particle size, raised by the power of approximately 1. As indicated above, vegetation will affect the transport capacity of runoff by controlling its volume and, through the effect on surface roughness, its velocity.

As an example of the type of calculations that can be made using the above equations, Table 2.11 lists some relative transport capacities calculated using the Engelund–Hansen equation for the runoff amounts given in Table 2.7. The values are approximate and are presented only to show the differences in transport capacity that may arise from differences in infiltration rates. The effects of soil properties on transport capacity are discussed in section 2.4.

In reality, the calculation of transport capacity is much more complicated because the sediment consists of a mixture of primary particles and soil aggregates of different sizes. For soils with a wide aggregate-size distribution, the single particle-size parameter included in the above equations is inadequate, and the transport capacity may have to be calculated separately for different particle or aggregate size classes. For a small amount of runoff, the area of 'attack' is limited to the area covered by small particles. When the runoff increases, more particle sizes and thus a larger surface area become accessible. When an increasing amount of fine material is removed from the surface and only the larger particles are left behind, hardly any material can be removed because the larger particles protect the soil surface. This effect is called armouring.

2.3.3 SOIL DETACHMENT BY FLOW

Soil detachment by flow is often considered as a function of the shear stress of the flow ($\tau = \rho_w g S Q$), raised to the third or fifth power, or as a linear function of the shear stress above a critical value (τ_c) which is related to the shear strength of the soil (τ_s). One example of an equation based on this view is:

Rose *et al.* (1983),

$$DF = 0.276\eta(\rho_w g S Q - \tau_s), \qquad (2.26)$$

where DF = the rate of soil particle detachment by flow, and η = the efficiency of bedload transport.

Alternatively, detachment is viewed as a function of the grain shear velocity (u^*g) above a critical value (u^*g_{crit}) which is dependent upon the cohesion of the soil. Total shear velocity (u^*) of the flow is defined by:

$$u^* = (gRS)^{0.5} \qquad (2.27)$$

and the grain shear velocity represents that portion which is associated with the roughness of the soil grains. The remaining shear velocity relates to microtopographic (form) roughness and the vegetation. The following equation is an example of this approach.

Rauws and Govers (1988):

$$DF = A(u^*g - u^*g_{crit}), \qquad (2.28)$$

where A = a coefficient expressing soil, including grain size characteristics, and

$$u^*g_{crit}(\mathrm{cm/s}) = 0.89 + 0.56c, \qquad (2.29)$$

where c = cohesion of the soil (kPa) at saturation as measured with a torvane.

Several authors (Meyer and Wischmeier, 1969; Foster and Meyer, 1975; Beasley, Huggins and Monke, 1980; Park, Mitchell and Scarborough, 1982) consider that the detached material fills up the transport capacity of the flow and that sedimentation occurs only when the flow is overloaded. Alternatively, it might be assumed that the detached material settles continuously and that the amount present in the flow represents a balance between detachment and sedimentation (Rose *et al.* 1983; Nielsen and Styczen, 1986). The net effect when the situation is viewed as a balance of processes is described by the following equation:

$$DF = \eta Qs_{EH}, \qquad (2.30)$$

where = η is the ratio of energy spent by the flow on lifting particles to the total amount of energy spent on lifting plus detachment; and Qs_{EH} = flow transport capacity calculated according to Engelund and Hansen (equation 2.21). Equation 2.26 (Rose *et al.* 1983) follows the same line of reasoning but is derived in a different way.

On the basis of laboratory tests with sandy, clay loam and clay soils and with sand, Quansah (1985) obtained the empirical relationship:

$$DF = e^{16.37}Q^{1.5}S^{1.44}d_{50}^{-1.54}, \qquad (2.31)$$

where DF is in kg/m^2, Q in m^3/m s and d_{50} in mm.

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These equations show that whether detachment by flow is described by itself or sedimentation is included, the quantity of runoff generated plays an important role. The effect of runoff, as measured by the value of the power exponent, is greatest when sedimentation is considered. The empirically derived value of 1.5 (equation 2.31) is closer to the value of 1.67 (equation 2.30) suggested when sedimentation is taken into account than to the value of 1.0 (equation 2.26) when it is not. Inclusion of sedimentation may therefore be the most promising approach.

As described earlier, differences in interception and infiltration due to vegetation will have an important influence on runoff generation and, therefore, on detachment by flow. Roughness is accounted for only in equation 2.30. According to this equation, doubling the roughness will decrease the amount of detachment by 20% for a given quantity of runoff.

2.3.4 TRANSPORT OF SPLASHED MATERIAL

Splash erosion may take place in the absence of runoff. However, the amount of material removed from a hillslope or a catchment in this manner is very small, even though it may be the dominant erosion process where runoff amounts are negligible.

When runoff occurs, the transport of the splashed particles depends upon the quantity of the splashed material which goes into the flow and on the flow velocity. Viewed in a very simple manner, splashed material, except for very small particles, behaves as suspended load and moves with the water until it settles. The average fall height will be a function of water depth, D (e.g. $D/2$, because the splashed material could be rather evenly distributed due to turbulence). The time (t) taken to settle becomes $D/2w$, where w is the average fall velocity of a particle of a given size and density. The distance moved by this particle becomes $t \cdot u = uD/(2w) = Q/(2w)$ (Styczen and Nielsen, 1989).

The distance upstream from which splashed material is transported over a given point thus varies with runoff quantity, particle size and particle density. The greater the velocity of runoff generated, the more splashed material is likely to be moved out of an area. The larger the amount of detachment, the greater will be the speed of sedimentation and the shorter the distance moved by the detached particle. Thus infiltration and surface roughness, through their effects on runoff generation, are important controls over the splash process.

However, if the soil surface is totally covered by water of some depth, splash does not take place at all. Relations suggested by Park, Mitchell and Scarborough (1982) and Rose *et al.* (1983) indicate that with a water depth of 6–8 mm and raindrops of 2.0–2.5 mm, splash is reduced by 80–90% compared with that for zero water depth. Equation 2.9 proposes that splash decays exponentially with increasing water depth. Vegetation in contact with the soil surface can increase water depths by reducing flow velocity.

2.3.5 SEDIMENTATION

Vegetation not only retards flow but acts as a filter to sediment being carried in the flow. The denser the vegetation, the more sediment can be trapped and removed from the flow. The effectiveness of close-growing vegetation in causing sedimentation has been modelled for grass barriers using laboratory experiments (Tollner, Barfield and Hayes, 1982; Hayes, Barfield an Tollner, 1984). The sediment wedge created by the barrier consists of three zones (Figure 2.14): (A) the surface slope, (B) the foreslope, and (C) the bottom slope. As the sediment is trapped and the sediment wedge builds up, these zones migrate downslope or downstream.

The model is most easily understood by starting with zone C and calculating its trapping efficiency (fC) over time (t):

$$fC(t) = \exp - 1.05 \times 10^{-3}(u_m \cdot R_s)/\nu)^{0.82} (w \cdot L(t)/u_m \cdot H)^{-0.91}, \qquad (2.32)$$

where $f(C) = (Q_{sd} - Q_{so})/Q_{sd}$; Q_{so} = sediment outflow from the filter; Q_{sd} = sediment trans-

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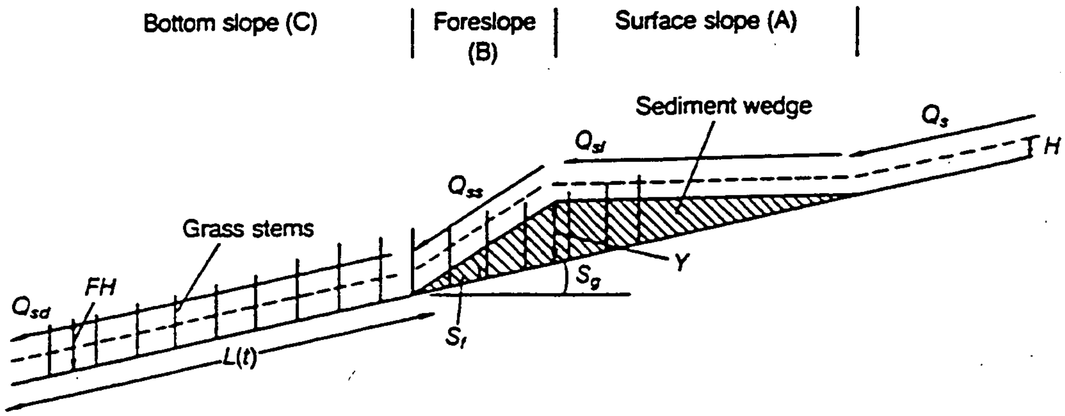

Figure 2.14 Schematic representation of sedimentation in a grass filter strip. Q = runoff; H = flow depth; Y = height of sediment wedge; FH = height of the grass in the filter; S_g = ground slope; S_f = foreslope of sediment wedge. Other notation described in the text (after Hayes, Barfield and Tollner, 1984).

port over zone C; $u_m = 1/nR_s^{2/3}S_g^{1/2}$; $R_s = (SS \cdot H)/(2H + SS)$; ν = kinematic viscosity of the water; w = settling velocity of the sediment; $L(t)$ = length of zone C along the slope; S_g = slope of the ground surface; SS = average spacing of the grass stems; H = depth of flow; and n = a modified Manning's n (S/cm$^{1/3}$) approximated as 0.012 for grass stems.

This relationship is based on an application of the Manning equation of flow velocity using an analogy between flow in a rectangular channel and flow between grass stems to give a channel with a width equal to the average spacing of the stems. The critical factors determining the efficiency of the filter are the density, shape and resilience of the grass stems as these affect the surface roughness. The hydraulic radius (R_s) is defined here as the spacing hydraulic radius.

The term Q_{sd} is estimated using an Einstein-type sediment transport equation:

$$\Delta(d/S_gR_s) = 1.08(Q_{sd}/(\gamma_s \Delta g d^3))^{-0.28}, \tag{2.33}$$

where $\Delta = (\gamma_s - \gamma_w)/\gamma_s$.

Knowing Q_{sd} and the right-hand side of equation 2.32, the sediment outflow (Q_{so}) and the trap efficiency (fC) are determined.

Sediment transport over zone B (Q_{ss}) is represented by:

$$Q_{ss} = (Q_{si} - Q_{sd})/2, \tag{2.34}$$

and the trapping efficiency of zone B (fB) is calculated from:

$$fB = (Q_{si} - Q_{sd})/Q_{si}. \tag{2.35}$$

The term Q_{ss} is estimated using equation 2.33 but substituting Q_{ss} for Q_{sd} and S_f (slope of the foreslope of the sediment wedge) for S_g.

Sediment transport over the surface slope of zone A is represented by Q_{si} and the mass deposition rate for the wedge is Q_{su}. Then

$$Q_{si} = Q_s - Q_{su}, \tag{2.36}$$

$$Q_{su} = (\gamma_{sdep}(Y_f^2 - Y_i^2))/(2(t_f - t_i)S_g), \tag{2.37}$$

$$Y_f(t) = FH \qquad \text{for } Y_f(t) \geqslant FH, \tag{2.38}$$

$$Y_f(t) = (2/\gamma_{sdep}(fB \cdot Q_{si} \cdot S_e)(t_f - t_i) + Y_i(t)^2)^{1/2} \qquad \text{for } Y_f(t) < FH, \tag{2.39}$$

where Qs = sediment transport upslope of the barrier; Y = height of the sediment wedge; $(t_f - t_i)$ = difference in time between time periods f and i; γ_{sdep} = unit weight of deposited sediment; S_e = angle between foreslope of the sediment wedge and the ground slope; and FH = height of the grass in the filter.

The extent of the sediment wedge upslope (Z_f) is calculated from:

$$Z_f = Y_f/S_g. \tag{2.40}$$

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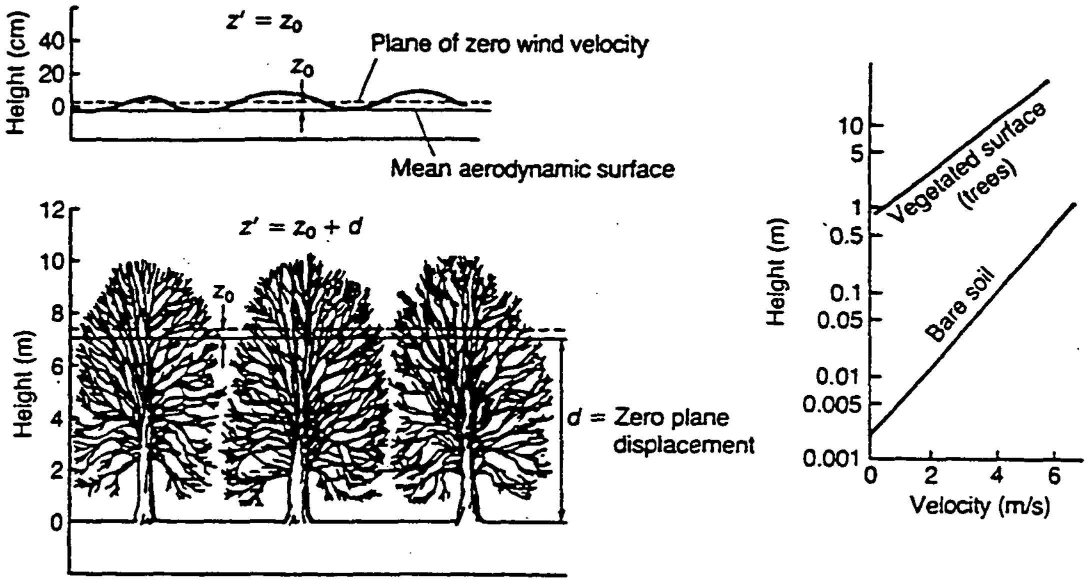

Figure 2.15 Wind velocity as a function of height above a bare and a vegetated soil surface.

The total amount of sediment trapped in the filter (T) is determined from:

$$T = Q_{su} + (Q_{sd} - Q_{so}) + (Q_{si} - Q_{sd}). \quad (2.41)$$

The trapping efficiency of the filter is:

$$f = (Q_s - Q_{so})/Q_s. \quad (2.42)$$

In addition to the properties of the grass stems mentioned above, the effectiveness of the filter will depend upon the height of the grass as this will influence the volume of sediment that can be trapped.

Sedimentation in the filter causes slope steepness to decline as the ground slope is replaced by the slope of zone A in the sediment wedge. Flow velocities are thus decreased and the erosive capacity of the flow reduced. However, the foreslope of the sediment wedge is steeper than the ground slope. Whilst the foreslope remains within the barrier, the potential increase in velocity over this steeper slope is largely offset by the roughness imparted by the grass stems. When the foreslope has migrated downslope to the edge of the barrier, however, flow leaving the barrier will have its velocity increased. Since, as shown in laboratory experiments by Emama (1988), most of the sediment originally carried in the flow is deposited within and upslope of the barrier, the flow is now largely sediment free and is therefore able to effect considerable erosion.

2.3.6 MODIFICATION OF AIR FLOW

Shear velocity in open ground

In the absence of convective eddies generated by vertical temperature gradients, wind speed over uniform level open ground increases logarithmically with height from a height (z_0) which is defined as the height above the mean aerodynamic surface at which wind velocity is zero (Figure 2.15). According to Bagnold (1941), the open field wind velocity profile is described by the equation:

$$u(z) = 2.3/k \cdot u^{*} \cdot \log(z/z_0), \quad (2.43)$$

where u = mean wind velocity at height z; k = von Karman universal constant for turbulent flow (= 0.4 for clear fluids); and u^{*} = drag or shear velocity.

The term z_0 is known as the roughness length and is a measure of the ground surface roughness. Bagnold (1941) found that z_0 was equal to

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Table 2.12 Drag coefficients for resistance of vegetation in moving air (after Wright and Brown, 1967; Randall, 1969; Voetberg, 1970; Morgan and Finney, 1987)

Cover	z_0 *(cm)*	*CD*	*Cd*
Grass	0.2–0.3	0.005–0.009	
Sugar beet	0.4–1.6		0.003–0.33
Wheat	1.2–3.0		0.001–0.08
Barley			0.001
Planted straw strips	2.1		0.001–0.08
Onions	0.8		0.006–0.50
Peas	0.4		0.001–0.23
Potatoes	5.4		0.001–0.07
Broad beans			0.01–0.05
Apple orchard (winter)		0.02–0.03	
Apple orchard (summer)		0.06–0.07	
Maize		0.01–0.10	0.02–0.15
Rice		0.01–0.10	
Coniferous forest	1.0	0.03–0.10	
Deciduous forest	1.8	0.01–0.03	

z_0, *CD* and *Cd* are defined in the text.

Values for *Cd*, except for maize, are for crop biomass in the lower 5 cm of the atmosphere and for a wind velocity at 5 cm height of 1 m/s.

The values of z_0, *CD* and *Cd* shown here are typical for the cover types specified. In practice, values of z_0 decrease with increasing wind velocity whereas those of *CD* and *Cd* can both increase and decrease with increasing wind velocity. Values of all three coefficients increase with increasing plant growth.

about 1/30th of the height of the sand particles or stones that caused the roughness. Other workers, however, indicate that z_0 approximates 1/10th of the height of the roughness elements (Monteith, 1973; Bache and MacAskill, 1984).

The shear velocity can be calculated by rearranging equation 2.43. It is thus directly proportional to the rate of increase in wind velocity with the logarithm of height. Since it is equivalent to the slope of the line in Figure 2.15, it can be determined by measuring the wind speed at two different heights, plotting the results on a graph of velocity versus the logarithm of height, joining the points with a straight line and calculating the slope of the line expressing the change in velocity for a unit change in log height. It should be noted that the convention of plotting the dependent variable on the *y*-axis gives way to the convention of plotting height vertically. Thus, higher shear velocities appear as gentler-sloping lines on the graph since they represent a high value of change in the *x*-axis for a unit change in the *y*-axis.

Shear velocity is not an actual velocity but has the same units as velocity. It is defined by

$$u^* = \tau/\rho_a, \qquad (2.44)$$

where τ = surface shear stress exerted by the air flow; and ρ_a = density of the air (= 0.00123 Mg/m^3 as an average value at sea-level).

Shear velocity with vegetation

Vegetation reduces the shear velocity of the wind by exerting a drag on the air flow. This is compensated for by a transfer of momentum from the air to the vegetation which, for an incompressible fluid, implies a reduction in velocity. Vegetation thus acts as a momentum sink. Vegetation increases the roughness length, z_0, which can be approximated in value as 1/10th of the height of the plant canopy. Typical values of z_0 for a range of surfaces are given in Table 2.12. A vegetation cover also

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displaces the height of the mean aerodynamic surface above the ground by a distance, d, known as the zero plane displacement (Figure 2.15). The value of d is usually approximated as 0.7 times the height of the plant canopy. Wossenu Abtew, Gregory and Borrelli (1989) have shown that a more accurate assessment can be obtained from $d = H \times F$, where H is the average height of the individual roughness elements and F is the fraction of the total surface covered by those elements. They also show that the roughness length can be estimated from $z_0 = 0.13\,(H - d)$. The term z_0 may be viewed as a measure of the bulk effectiveness of the vegetation cover in absorbing momentum, and the term d is a measure of the mean height at which the absorption takes place (Thom, 1975).

Drag coefficients

The frictional drag exerted on the atmosphere by a vegetation canopy in bulk can be expressed by an equation derived from a simplification of the Navier–Stokes equation for the conservation of momentum in incompressible, steady, two-dimensional air flow (Seginer, 1972; Skidmore and Hagen, 1977; Hagen *et al.*, 1981):

$$\tau = 1/2\rho_a u(z)^2 CD, \qquad (2.45)$$

where τ = the drag force per unit horizontal area of vegetation; and CD = a bulk drag coefficient.

Equating the expressions of τ in equations 2.44 and 2.45 gives:

$$CD = 2u^{*2}/u(z)^2. \qquad (2.46)$$

Considering this equation alongside equation 2.43, it is clear that a relationship exists between the bulk drag coefficient and the aerodynamic properties of the crop canopy as expressed by z_0 and d. In general terms, the rougher the surface and the higher the zero plane displacement, the greater is the drag coefficient. Since both z_0 and d vary with vegetation type and its stage of growth, the drag coefficient, CD, will also vary. All three terms are also dependent upon wind speed. As wind velocity increases, CD and z_0 should fall as a result of streamlining of the foliage elements downwind and d will fall because of the greater penetration of wind into the canopy. The drag coefficient is therefore dynamic and cannot be represented by a single value as is normally the case with a rigid body. Typical values are given in Table 2.12.

Contrary to the above, the bulk drag coefficient has been found by Randall (1969) in apple orchards and Bache (1986) with cotton canopies to increase with increasing wind speed. This may be explained by the dependence of the above on the assumption that the whole leaf area contributes to the momentum transfer whereas, in reality, the effective foliage area for momentum absorption changes in a complex manner through streamlining, leaf flutter and plant vibration as the wind speed alters. In such cases, the terms CD, z_0 and d, which express the effect of the vegetation in bulk, are only broad and not necessarily truthful indicators of what is happening. Also, their emphasis on conditions at the interface between the plant canopy and the atmosphere above, rather than on the soil surface beneath the vegetation, limits their value for determining the likelihood of wind erosion occurring.

An alternative and arguably more meaningful approach is to derive a drag coefficient (Cd) to describe the effect of the vegetation on the air within the plant layer. This is achieved by balancing the drag force of the wind profile exerted on the vegetation at height (h) with the extraction of momentum due to the frictional surface area of the individual foliage elements. This gives (Wright and Brown, 1967):

$$\tau(h) = 0.5\int_0^h Cd\,A(z)\,u(z)^2\,dz, \qquad (2.47)$$

where $A(z)$ is the leaf area density (i.e. leaf area per unit volume).

Substituting equation 2.44 and rearranging yields:

$$Cd = 2u^{*2}\Big/\int_0^h u^2 A(z)\,dz. \qquad (2.48)$$

Wright and Brown (1967) found that values of Cd for maize leaves varied with height within

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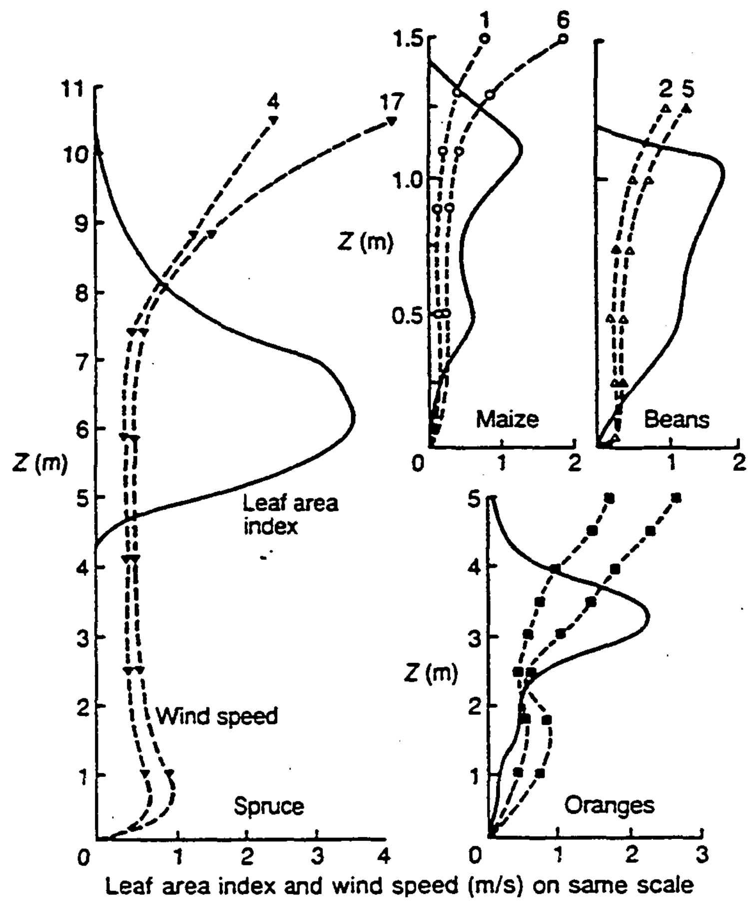

Figure 2.16 Wind velocity as a function of height and leaf area index for four crop types (after Landsberg and James, 1971). Two profiles are shown for each crop.

the canopy and increased with increasing wind velocity. These results emphasize the importance of variations in vegetation structure and effective foliage area in controlling the amount of drag and, thereby, the form of the wind profile.

Figure 2.16 shows how wind velocity changes with height within the vegetative layer for a range of vegetation or crop types as a function of leaf area density (Landsberg and James, 1971). Equation 2.43 is only valid as an expression of the velocity profile in the air above the zero plane displacement. Below this, the wind profile may be fitted by one of the following equations:

$$u(z) = u(h)\,(1 + m(1 - z/h))^{-2} \quad (2.49)$$

$$u(z) = u(h)\exp(-n(1 - z/h)), \quad (2.50)$$

where m is an experimental parameter which is characteristic of the vegetation type (Thom, 1971) and n is an attenuation coefficient which typically varies between 2 and 5 in value depending upon the foliage density and the type of vegetation (Inoue, 1963; Cionco, 1965).

Equations 2.49 and 2.50 are normally valid with tall vegetation such as trees. With shorter vegetation, for example, maize and rice, equation 2.50 describes the wind profile in the top half of the plant layer reasonably well but in the lower 20% of the profile the wind speeds decrease more slowly than the equation predicts (Denmead, 1976). Indeed, they may even increase close to the ground surface because of the sparser vegetation cover at the bottom of the plant layer. Wind tunnel studies (Morgan, Finney and Williams, 1986) on crops less than

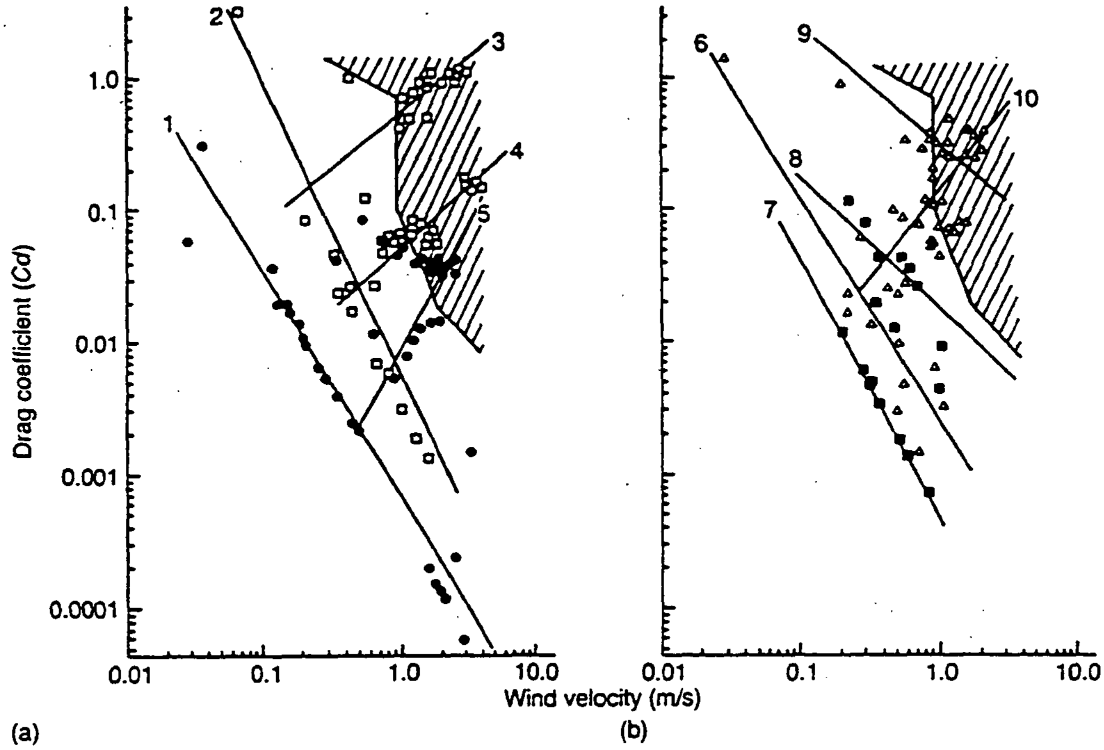

Figure 2.17 Relationships between drag coefficients (*Cd*) for the 5 cm layer of vegetation close to the ground surface and wind velocity measured at 5 cm height for (a) planted straw strips (●) and onions (□); and (b) live barley strips (■) and sugar beet (△). Shaded area represents risk of soil particle movement (from Morgan, 1989.)

0.15 m tall showed that the wind profile within the vegetation layer was better described by the equation:

$$u(z)/u(h) = 0.9 + 0.1 \ln (z/h). \quad (2.51)$$

Shaw and Pereira (1982) and Hagen and Lyles (1988) also apply a log height-velocity relationship to the air in the lowest part of a vegetation cover close to the ground surface.

Morgan and Finney (1987) attempted to examine conditions in the lower part of the atmosphere by using equation 2.48 to calculate drag coefficients for the vegetation in the lowest 5 cm of single crop rows from field measurements of average 10 s wind velocities made with cup anemometers. They found that the drag coefficients both increased and decreased in value with wind speed (Figure 2.17), depending upon the consistency of the wind. If the latter is expressed by an index of turbulence (TU), defined as the ratio of the values of the standard deviation to the mean of a series of consecutive wind velocity recordings, the drag coefficient decreases with increasing wind speed when $TU > 0.2$. For $TU \leqslant 0.2$, the drag coefficient increases with wind speed, presumably as a result of leaf flutter in the more continuous wind disturbing the atmosphere surrounding the foliage and setting up a 'wall' effect.

Although the correlation coefficient (r) between values of the drag coefficient and wind speed was always higher than -0.80 ($P < 0.02$) for the negative relationship, it was generally lower for the positive one. This implies that wind speed alone does not adequately explain the variability in drag coefficients. Further insight is provided by wind tunnel studies (Morgan, Finney and Williams, 1988) of the drag coefficient of individual leaves (CL) as a function of their morphological properties of size, shape, fragmentation, orientation and rigidity. These showed that the single-leaf drag coefficient increased as the projected area and the deflection angle decreased and the down

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wind alignment increased. The results imply that highest drag is associated with greatest contact length between the wind and the air flow in a downwind direction and not with the area of foliage facing the wind; a finding which points to the importance of flow separation around the leaf and skin friction from the leaf surface over form drag in contributing to wind resistance. Since bladed leaves were found to have lower deflection angles than round or ovate leaves, it follows that high drag is associated with bladed leaves aligned downwind not as a result of streamlining but because of their natural growth position.

Information on foliage properties can add substantially to the understanding of drag coefficient (Cd) values in the lowest 5 cm of the atmosphere. From the field data of Morgan and Finney (1987), two relationships are obtained:

$$\log Cd = -1.648 - 1.406 \log u - 378.4\, PA + 0.00466\, H + 0.01045\, V \quad \text{for } TU > 0.2 \quad (R = 0.839;\ n = 159), \tag{2.52}$$

$$\log Cd = -0.139 + 0.316 \log u - 369.1\, PA + 0.1167\, BM - 1.757\, TU \quad \text{for } TU \leqslant 0.2 \quad (R = 0.727;\ n = 130), \tag{2.53}$$

where u = wind velocity (m/s) at 5 cm height; PA = projected area of the foliage facing the wind (m^2); H = average angle of the leaves from the vertical in a downwind direction (degrees); V = average angle of the leaves from the vertical in a crosswind direction (degrees); BM = biomass (kg DM/m^3); and TU = turbulence index (described above).

The terms PA, H, V and BM are determined for a representative 10 cm length of a plant row in the lowest 5 cm of the atmosphere. Both equations show that the drag coefficient increases as the projected foliage area facing the wind decreases, again demonstrating the importance of contact length downwind between the leaf surfaces and the air. This is also implicit in the positive effects of downwind leaf alignment and increasing biomass, the latter being another indication of greater surface area of foliage. Typical values for Cd are given in Table 2.12.

This analysis shows that plant properties can have an important influence over values of the drag coefficient (Cd). This, in turn, as combining equations 2.48 and 2.43 shows, has an effect on shear velocity:

$$u^* = 0.71 \int_0^h (CdA(z)\, u(z)\, dz)^{0.5}. \tag{2.54}$$

2.4 MECHANICAL EFFECTS

2.4.1 SOIL REINFORCEMENT

The roots and rhizomes of the vegetation interact with the soil to produce a composite material in which the roots are fibres of relatively high tensile strength and adhesion embedded in a matrix of lower tensile strength. The shear strength of the soil is therefore enhanced by the root matrix.

Field studies of forested slopes (O'Loughlin, 1984) indicate that it is the fine roots, 1–20 mm in diameter, that contribute most to soil reinforcement and that the larger roots play no significant role. Grasses, legumes and small shrubs can have a significant reinforcing effect down to depths of 0.75–1.5 m. Trees have deeper-seated effects and can enhance soil strength to depths of 3 m or more depending upon the root morphology of the species (Figure 2.18; Yen, 1972). Root systems lead to an increase in soil strength through an increase in cohesion brought about by their binding action in the fibre/soil composite and adhesion of the soil particles to the roots. It is generally held that roots have no effect on soil friction angle but Tengbeh (1989) found that grass roots increased the angle of internal friction of a sandy soil but had no such effect on a sandy clay loam.

The pattern of the relationship between soil cohesion and the roots is not known. Tengbeh (1989) found that Loretta grass (*Lolium perenne*)

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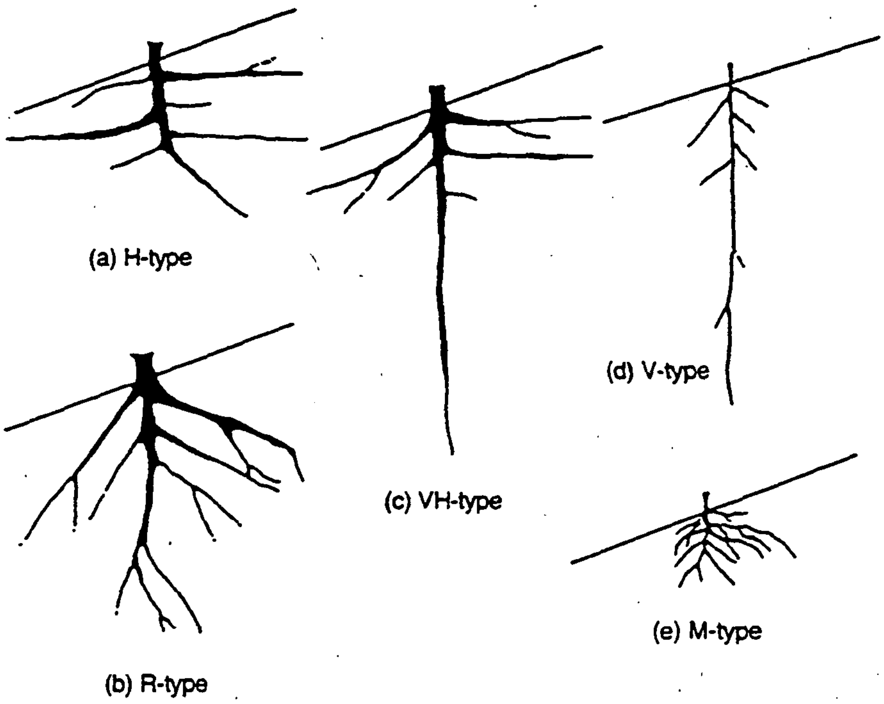

Figure 2.18 Patterns of root growth in trees (after Yen, 1987). (a) H-type: maximum root development occurs at moderate depth, with more than 80% of the root matrix found in the top 60 cm; most of the roots extend horizontally and their lateral extent is wide. (b) R-type: maximum root development is deep, with only 20% of the root matrix found in the top 60 cm; most of the main roots extend obliquely or at right angles to the slope and their lateral extent is wide. (c) VH-type: maximum root development is moderate to deep but 80% of the root matrix occurs within the top 60 cm; there is a strong tap root but the lateral roots grow horizontally and profusely, and their lateral extent is wide. (d) V-type: maximum root development is moderate to deep; there is a strong tap root but the lateral roots are sparse and narrow in extent. (e) M-type: maximum root development is deep but 80% of the root matrix occurs within the top 30 cm; the main roots grow profusely and massively under the stump and have a narrow lateral extent. H- and VH-types are considered beneficial for slope stabilization and wind resistance. H- and M-types are beneficial for soil reinforcement. The V-type is wind resistant.

increased the cohesion (c; kPa) of soil as a function of root density (RD; $\mathrm{Mg/m^3}$) in an exponential relationship so that for a sandy clay loam soil:

$$c = 10.54 + 8.63 \log RD \qquad r = 0.99,\ n = 16 \tag{2.55}$$

and for a clay soil:

$$c = 11.14 + 9.9 \log RD \qquad r = 0.99,\ n = 11. \tag{2.56}$$

In contrast, linear relationships have been obtained by Waldron (1977) between the change in soil shear strength and the root area ratio of barley roots in a silty clay loam soil, and by Ziemer (1981) between shear strength and roc biomass of *Pinus cordata* in a sand.

These studies show that root reinforcement can make significant contributions to soil strength, even at low root densities and low shear strengths. Equations 2.55 and 2.56 indicate that cohesion increases rapidly with increasing root density at low root densities but that increasing root density above 0.5 $\mathrm{Mg/m^3}$ on the clay soil and 0.7 $\mathrm{Mg/m^3}$ on the sandy clay loam soil has little additional effect. This implies that vegetation can have its greatest effect close to the soil surface where the root density is generally highest and the soil is otherwise weakest.

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Since shear strength affects the resistance of the soil to detachment by raindrop impact (Cruse and Larson, 1977; Al-Durrah and Bradford, 1982), and the susceptibility of the soil to rill erosion (Laflen, 1987; Rauws and Govers, 1988) as well as the likelihood of mass soil failure, root systems can have a considerable influence on all these processes. The maximum effect on resistance to soil failure occurs when the tensile strength of the roots is fully mobilized and that, under strain, the behaviour of the roots and the soil are compatible. This requires roots of high stiffness or tensile modulus to mobilize sufficient strength and the 8–10% failure strains of most soils. The tensile effect is limited with shallow-rooted vegetation where the roots fail by pullout, i.e. slipping due to loss of bonding between the root and the soil, before peak tensile strength is reached (Waldron and Dakessian, 1981). The tensile effect is most marked with trees where the roots penetrate several metres into the soil and their tortuous paths around stones and other roots provide good anchorage. Root failure may still occur, however, by rupture, i.e. breaking of the roots when their tensile strength is exceeded. The strengthening effect of the roots will also be minimized in situations where the soil is held in compression instead of tension, e.g. at the bottom of hillslopes. Root failure here occurs by buckling.

2.4.2 ROLE OF ORGANIC MATTER

The return of vegetative material to the soil as organic matter plays a vital role in aggregation of the soil particles. Aggregate-stabilizing compounds are formed during the degradation of organic material, such as manure, plant roots, leaves and stems, and straw, by microbial and faunal activity within the soil. Thus, the level of biological activity or the speed of degradation of organic matter are probably better indicators of the relative stability of soil aggregates than the content of organic matter as such. A good vegetative cover is likely to increase the biological activity and the rate of aggregate formation, but no quantification of this effect can be given. Increased aggregate stability of a soil increases permeability and infiltration which, in turn, reduces surface runoff and enhances the available water content for plant growth. This promotes better vegetation growth with greater protection of the soil surface and a drier soil environment.

The size and stability of the soil aggregates affects their detachability by raindrop impact and their detachability and transportability by surface runoff. Several of the transport equations (section 2.3.2) contain factors of the type d^{-x}, where x varies between 1.0 and 1.54, showing that transport capacity becomes inversely related to the diameter of the aggregates. As the Engelund–Hansen sediment transport relationship (equation 2.21) indicates, transportability is also dependent on the density of the soil particles. Thus, large soil aggregates are moved before primary particles of the same diameter because their densities are approximately 1.8–2.0 Mg/m^3 compared with 2.6 Mg/m^3 for sand.

2.4.3 ROOT WEDGING

Root wedging is a potentially destabilizing process whereby fissures and joints in rocks are opened up by the advance and growth of roots. Trees create the biggest problem, though grass roots can also force open small cracks. Where vegetation gains a hold on steep slopes with steeply-inclined joint planes or fissures, the wedging action of plant roots can dislodge and topple blocks or sections of the rock. Earth (soil) slopes are less likely to be affected.

Root wedging may not cause instability during the lifetime of a tree as the rocks may be enveloped within the roots and trunk. It is on the death of the tree that dislodged blocks are likely to fall free.

2.4.4 ARCHING AND BUTTRESSING

The tap and sinker roots of many tree species extend through the soil layers and into the

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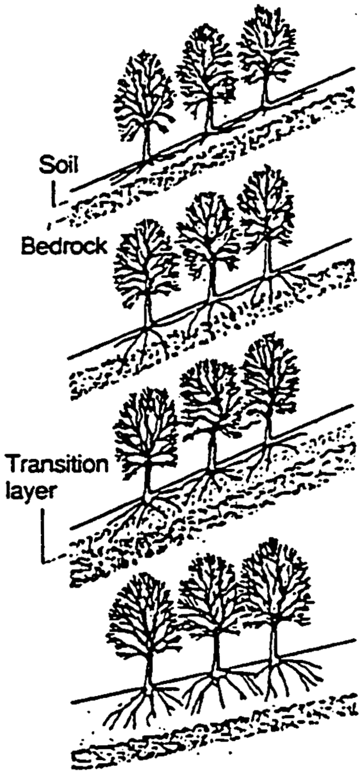

Slope type	Description	Stabilizing effect of roots
	A. Relatively thin soil mantle, fully reinforced with tree roots, underlain by massive bedrock impenetrable to roots	Slight – plane of weakness occurs at bedrock interface
	B. Similar to type A, except bedrock contains discontinuities which are penetrable by roots; trunks and roots act in the same way as toe piles	Major
	C. Thicker soil mantles containing a transition layer with soil density and shear strength increasing with depth; roots penetrating the transition layer provide a stabilizing force to the slope	Substantial
	D. Thick soil mantles below root zone; trees floating on unrooted underlying soil	Little effect on deep-seated instability

Figure 2.19 Classes of plant-root reinforced and anchored slopes (after Tsukamoto and Kusakabe, 1984).

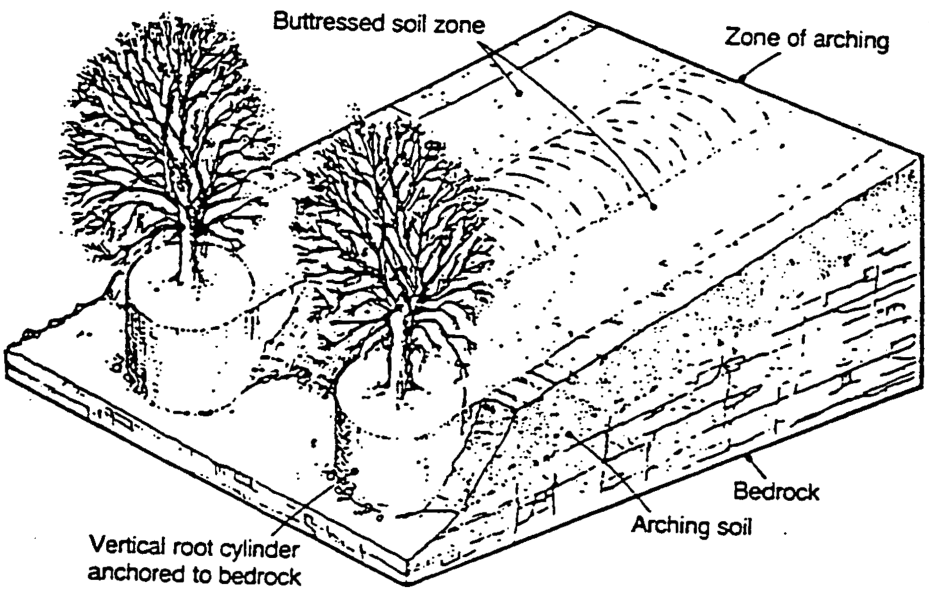

Figure 2.20 Schematic representation of soil buttressing and arching (after Wang and Yen, 1974).

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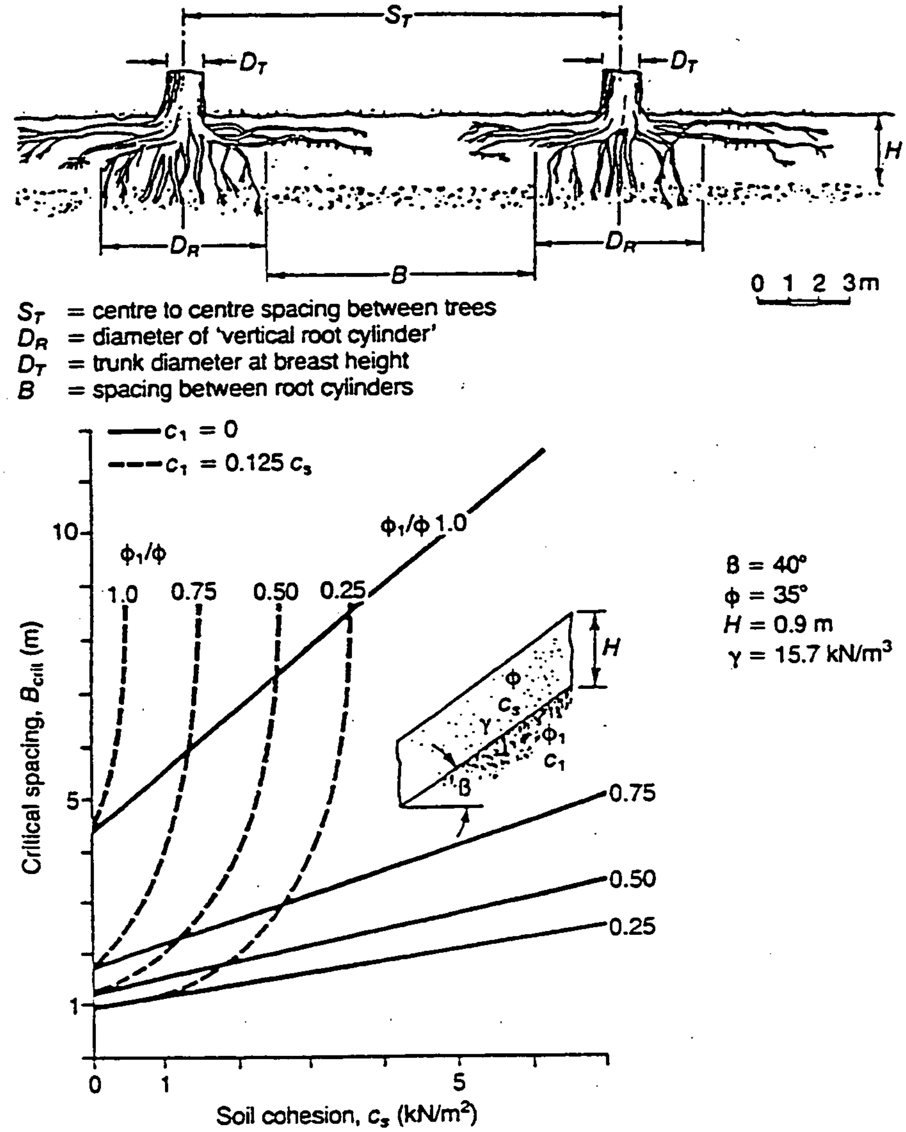

Figure 2.21 Critical spacing of trees for arching based on the theory of Wang and Yen (1974) applied to a steep sandy slope (after Gray and Leiser, 1982).

underlying bedrock, anchoring them to the slope. The trunks and large roots then act in the same way as stabilizing piles and buttress the soil, restraining it from movement downslope. The extent to which buttressing can contribute to the stability of the soil mass on a slope depends upon the depth of the soil mantle and the groundwater as well as on the penetrability of the bedrock by roots (Figure 2.19; Tsukamoto and Kusakabe, 1984).

Where trees are sufficiently close together, the soil between the unbuttressed parts of the slope may gain strength by arching (Figure 2.20). Based on work by Wang and Yen (1974), Gray (1978) has produced a plot of the theoretical critical (minimum) spacing required for arching to occur on a 40° slope with a 0.9 m deep sandy soil mantle (Figure 2.21). This shows that the critical spacing depends upon the cohesiveness of the underside of the supported soil mass. If cohesion is zero and the residual friction along the underside is half the peak friction, the critical spacing is only 1.2 m. If a cohesion of 2.4 kN/m^2 is assumed, with a residual

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cohesion of 12.5% of this value, the critical spacing increases to 6.4 m. Tree spacings on such slopes in the field are often of the right order for arching to develop.

2.4.5 SURCHARGING

Surcharge arises from the additional weight of the vegetation cover on the soil. This effect is normally considered only for trees, since the weight of grasses and most herbs and shrubs is comparatively small. Surcharge increases the downslope forces on a slope, lowering the resistance of the soil mass to sliding, but it also increases the frictional resistance of the soil. Bishop and Stevens (1964) show that large trees can increase the normal stress on a slope by up to 5 kN/m² but that no more than half contributes to an increase in shear stress. Generally, the second effect outweighs the first, so that, overall, surcharge is beneficial. Nevertheless, surcharge at the top of a slope can reduce overall stability whereas, at the bottom of the slope, it will increase stability.

De Ploey (1981) invokes surcharge combined with lowering of the cohesion of the soil mass through increased infiltration and, therefore, increased soil water content, as contributing to landslides on the forested slopes of the Serra do Mar, east of Santos, in Brazil. The surcharge becomes critical when rainfall of several hundreds of millimetres occurs in a wet spell of a few days; for example, on 17 and 18 March 1967 when daily rainfalls totalled 260 and 420 mm respectively. In such events, interception and evapotranspiration are reduced virtually to zero and the soil is unable to either dry out or drain. The critical factors here may well be the low angle of internal friction of the soil material which, when close to waterlogging, is reduced to less than 20°, and the steepness of the slopes, which are over 20°. In contrast, Gray and Megahan (1981) state that surcharge is beneficial when cohesion is low and groundwater levels are high provided that the angle of internal friction of the soil is also high and the slope angles are low.

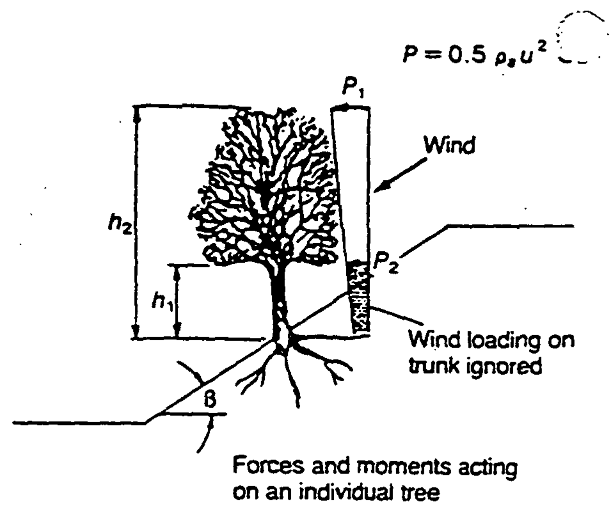

Figure 2.22 Effect of wind loading on a single tree (from Coppin and Richards, 1990).

2.4.6 WIND LOADING

The pressure (P) exerted on a vegetation cover by wind can be transmitted to the soil as an increased loading (D), reducing its resistance to failure. From the work of Hsi and Nath (1970) and Brown and Sheu (1975), for a single tree (Figure 2.22):

$$D = \sum_{i=h1}^{i=h2} (0.5\rho_a u^2 CD \cos^2 \beta b)\, i, \qquad (2.57)$$

where D = drag force (kg) transmitted into the slope; $h1$ = height of the bottom of the tree canopy above the ground (m); $h2$ = height of the top of the tree canopy above the groun[d]; ρ_a = density of the air (kg m³); u = wi[nd] velocity (m/s); CD = the bulk drag coefficient of the vegetation; β = slope angle (degrees); and b = transverse width of the crown (m) at each height increment, i.

Wind loading is only significant for trees and when the wind velocity exceeds 11 m/s.

Wind pressure on a tree can also produce a destabilizing moment which, if the tree is not well anchored, will cause it to topple over. Increased infiltration of water into the soil through the scar created by the uprooted tree can then lower the resistance of the whole soil mass to failure.

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2.4.7 SURFACE PROTECTION

Vegetation protects the soil mechanically by absorbing directly the impact of walkers, livestock and vehicles. Most studies of this effect have concentrated on the resistance of vegetation to damage by walking. When an individual walks over the ground surface, the soil and vegetation are compacted in the early part of each step under the pressure of the heel; at the end of the step, they are sheared by the movement of the toe. The shearing action is the most damaging (Quinn, Morgan and Smith, 1980).

Broadly, grasses are reasonably resistant and can withstand between 1000 and 2000 passes by walkers before the density of cover falls below 50%. In contrast, alpine plant communities can withstand about 60 passes and arctic tundra communities only eight passes (Liddle, 1973).

The effect of walking on shrubs is greater than that on grasses because plants that produce buds and shoots at or below ground level have their growth points protected by the overlying foliage as the vegetation is flattened underfoot. They are therefore less easily damaged. For this reason, heath and bracken disappear more rapidly than grasses in upland areas under heavy recreational use.

2.5 VISUALIZATION

In order to visualize the combined effect when several vegetation-related parameters are varied simultaneously, some simple calculations are presented. The numerical values should not be taken too seriously, as the overall results only illustrate the theoretical principles previously discussed. Nevertheless, the influence of vegetation is demonstrated to about the right order of magnitude.

2.5.1 WATER EROSION

The effect of vegetation cover on water erosion is illustrated for seven conditions for which typical values of percentage cover and plant height are presented in Table 2.13. A value of saturated hydraulic conductivity has been chosen for a loamy soil and then varied, on the basis of McKeague, Wang and Coen (1986), taking account of the number of biopores and the level of soil aggregation expected under each condition. Values of Manning's n are selected from Engman (1986). The soil is assumed wet, so the infiltration rate is regarded as equal to the saturated hydraulic conductivity.

The accumulated runoff water is calculated for four rainfall intensities, three slope lengths or distances downslope, and two slope steepnesses. For each combination, the water depth, flow velocity and, using the Engelund and Hansen (1967) formula, transport capacity relative to that on bare soil are determined. Soil detachment by raindrop impact is calculated relative to that for bare soil using the procedure of Styczen and Høgh-Schmidt (1988). None of the calculations take into consideration an

Table 2.13 Parameters used for assessment of the effect of vegetation on erosion by water

Cover type	*Saturated hydraulic conductivity (mm/h)*	*Manning's n*	*Percentage cover*	*Height (m)*
Bare soil	10	0.01	0	
Grass	50	0.20	90	0
Soya beans	25	0.04	80	0.5
Maize	25	0.02	80	1.5
Agricultural crops and crusted soil	5	0.02	80	1.5
Eucalyptus and crusted soil	5	0.01	80	3–8
Eucalyptus with grass	50	0.20	80–90	3–8/0

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aggregate size distribution of the soil. A crusting index (Table 2.14) is developed based on the energy of the rainfall received at the ground relative to that received by a rain of 25 mm/h intensity on a bare soil. A rilling index (Table 2.15) is calculated as the product of runoff volume, slope length and slope steepness.

Table 2.14 Crusting index calculated as a function of rainfall energy. The index unit is the energy received relative to that on a bare soil when the rainfall intensity is 25 mm/h

	Rainfall intensity (mm/h)			
Cover	*25*	*50*	*75*	*100*
Bare soil	1	2.2	3.6	5.0
Grass, 90% cover	0.1	0.2	0.4	0.5
Soya beans	0.5	1.0	1.5	2.1
Maize	1.1	2.0	3.2	4.3
Eucalyptus, crusted soil	2.8	5.4	7.9	10.4
Eucalyptus with grass	0.3	0.5	0.8	1.0

In Figure 2.23, curves representing relative transport capacities and soil detachment by raindrop impact are presented for each condition. According to the value chosen for soil erodibility, the curves for soil detachment may be multiplied by a constant and the cross-over points of the two curves will change accordingly. Detachment by runoff has not been added as its relative importance depends upon erodibility. It is, however, probably proportional to the transport capacity. The curves for soil deta... ment differ from those presented in a sim...

Table 2.15 Rilling index calculated as the product of runoff, slope length and slope steepness for a 5% slope and different rainfall intensities

	Rainfall intensity (mm/h)			
Cover type	*25*	*50*	*75*	*100*
Slope length 20 m				
Bare soil	0.15	0.4	0.65	0.9
Grass	0	0	0.25	0.5
Soya beans	0	0.25	0.5	0.5
Maize	0	0.25	0.5	0.75
Agricultural crops with crusted soil	0.2	0.45	0.7	0.95
Eucalyptus with crusted soil	0.2	0.45	0.7	0.95
Eucalyptus with grass	0	0	0.25	0.5
Slope length 50 m				
Bare soil	0.38	1.0	1.63	2.25
Grass	0	0	0.63	1.25
Soya beans	0	0.63	1.25	1.88
Maize	0	0.63	1.25	1.88
Agricultural crops with crusted soil	0.5	1.13	1.75	2.38
Eucalyptus with crusted soil	1.0	1.13	1.75	2.38
Eucalyptus with grass	0	0	0.63	1.25
Slope length 100 m				
Bare soil	0.75	2.0	3.25	4.5
Grass	0	0	1.25	2.5
Soya beans	0	1.25	2.5	3.75
Maize	0	1.25	2.5	3.75
Agricultural crops with crusted soil	1.0	2.25	3.5	4.75
Eucalyptus with crusted soil	1.0	2.25	3.5	4.75
Eucalyptus with grass	0	0	1.25	2.5

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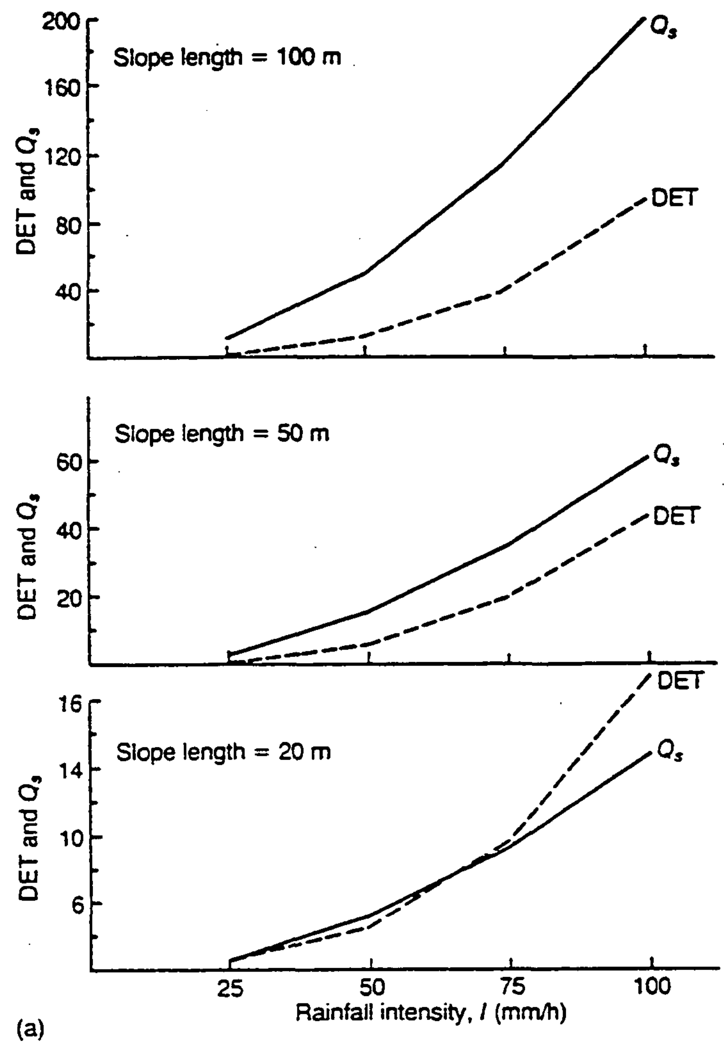

Figure 2.23 Relative transport capacities (Q_s) and soil detachment (DET) by splash as a function of rainfall intensity on a 5% slope for slope lengths of 100 m, 50 m and 20 m on (a) bare soil; (b) grass with and without a cover of Eucalyptus; (c) soya beans; (d) maize; (e) soya beans and maize on a crusted soil; and (f) bare soil with and without a crust under Eucalyptus. Note that the scales of the *y*-axis differ for the three slope lengths.

exercise by Foster and Meyer (1975) because the procedure used here relates the splashed material transported away to the quantity of the runoff instead of assuming that all the splashed particles are available for transport.

At the top of the slope (represented by the 20 m slope length), where only small amounts of runoff occur, the erosion is transport-limited except under grass. Further downslope (50 m slope length), where more runoff accumulates, the transport capacity exceeds soil detachment and the erosion becomes detachment-limited for conditions with vegetation close to the surface and for bare soil. At the foot of the slope, after 100 m slope length, erosion is detachment-limited except for trees without undergrowth and litter. However, even though erosion remains transport-limited under trees, the large amounts of runoff water accumulated may cause rilling to occur. In this case, detachment in the rills may dominate the erosion completely. As the transport capacity and the erosive capacity of rill flow are much larger than for sheet flow, much more material can be expected to be

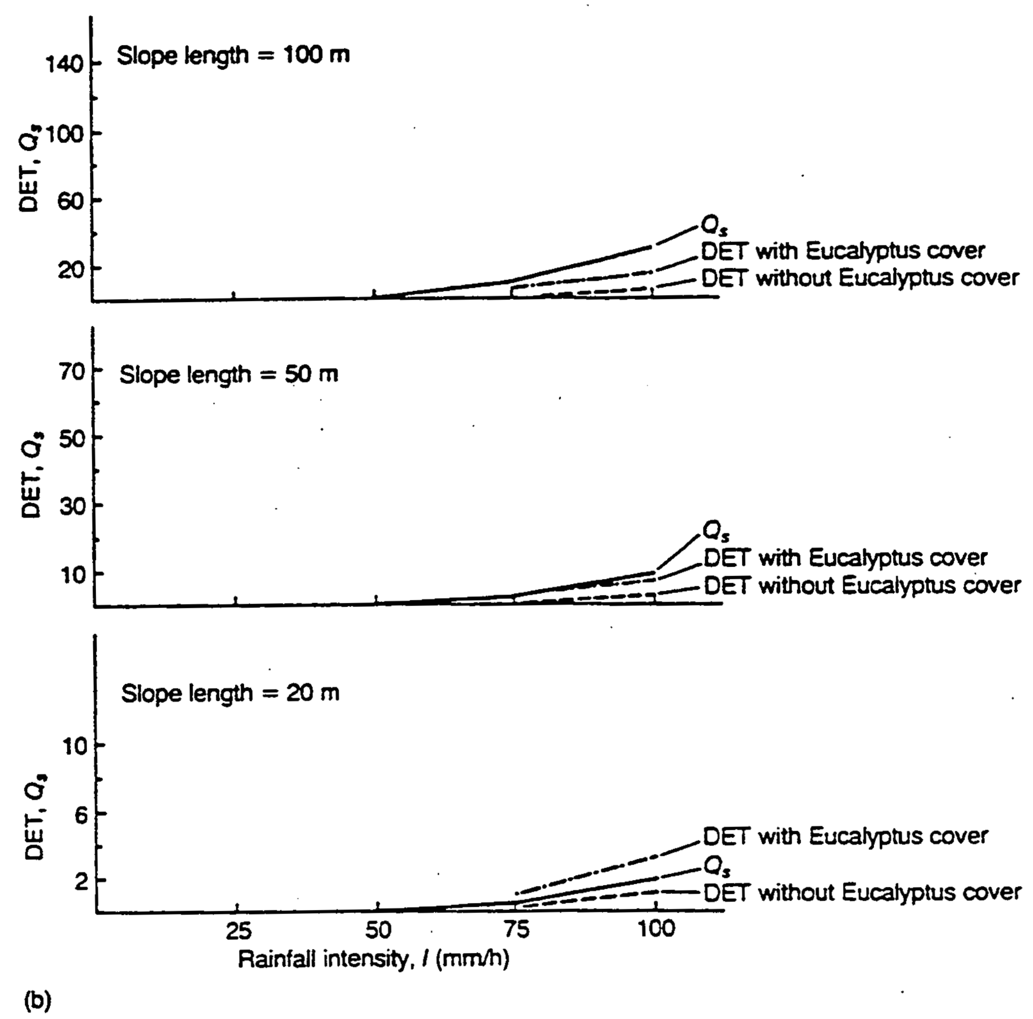

Figure 2.23 (cont.)

removed from an area when rills form. Passing the threshold for rill formation means a serious acceleration in water erosion.

A major difference between the curves is the starting point at which runoff commences. This hydrological effect may be just as important in determining the magnitude of erosion as the effects of vegetation on runoff volume.

The difference in the transport capacity for maize and soya bean is due to differences in surface roughness, while the difference in soil detachment by raindrop impact is due to the different plant heights. The curves for grass show a very low amount of runoff generated, a high degree of roughness and almost no detachment.

The most serious condition is for the trees without cover. This is because of the large amounts of runoff which are generated v[illegible] quickly after the onset of the storm, due [illegible] high values of the crusting index, and the high flow velocities. Despite the large amounts of runoff, soil loss is transport-limited, indicating extremely high detachment rates which arise from the high fall heights of the leaf drainage. The rilling index is larger here than in any of the other cases.

Two studies which support the validity of the last case have been carried out in Java, Indonesia, by Coster (1938) and Wiersum (1985). Both authors studied the effects of various vegetation layers in an *Acacia auriculiformis* forest on surface erosion by removing

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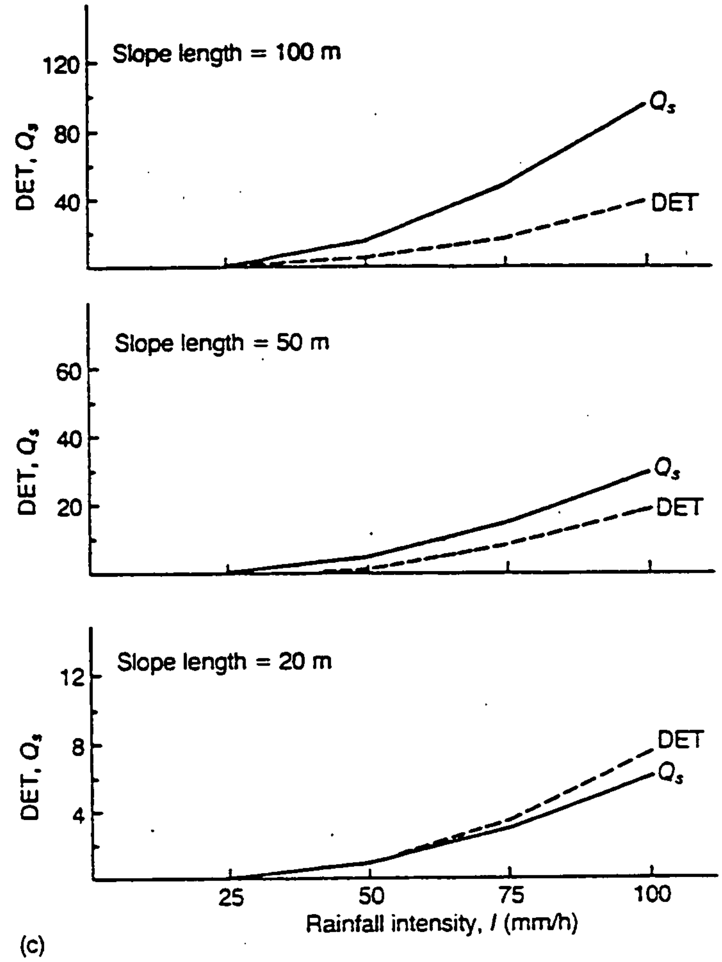

Figure 2.23 (cont.)

one vegetation layer at a time. Their findings clearly illustrate the importance of the height of the canopy above the soil surface and the importance of a ground vegetation or litter cover.

Coster (1938; Table 2.16) found that far more erosion occurred under trees without undergrowth and litter than on bare soil. When the trees were removed but litter and undergrowth kept intact, the soil loss was only 1/15th of that from bare soil. In the undisturbed forest, the soil loss was very low. Wiersum (1985) obtained less drastic differences but at least 20 times as much soil was lost when litter was removed compared to when litter was present. When both litter and undergrowth are intact, hardly any erosion occurs.

In both cases, it is the vegetation close to the soil surface and the litter that play the important role in controlling the erosion. Although the vegetation layers in the canopy catch rainwater and divert some to stemflow, these effects are more than offset by the increase in drop size of the rain which reaches the ground surface as leaf drainage.

2.5.2 SLOPE STABILITY

The stability of a slope against failure is evaluated by the factor of safety (F), which is defined as the ratio of the resistance of the soil mass to shear along a potential slip plane to the shear force acting on that plane. Soil failure occurs when the ratio falls to unity. The simple case of a translational failure along a sliding surface

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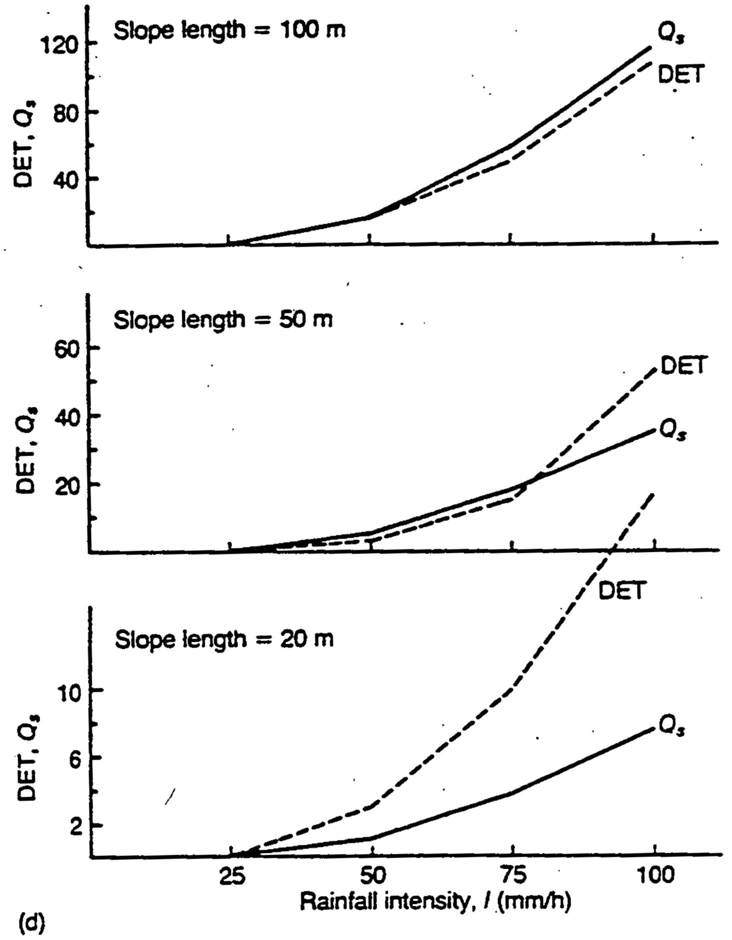

Figure 2.23 (cont.)

parallel to the ground over a relatively long uniform slope can be analysed by infinite slope analysis. In this case, a single element or segment (Figure 2.24) of the slope can be considered as representative of the whole, and the head and toe portions of the slope are ignored as being negligible in extent.

Using effective stress analysis, the factor of safety without vegetation can be defined by:

$$F = \frac{c' + (\gamma z - \gamma_w h_w) \cos^2 \beta \tan \phi'}{\gamma z \sin \beta \cos \beta}, \tag{2.58}$$

where c' = effective soil cohesion (kN/m^3); γ = unit weight of soil (kN/m^3); z = vertical height of soil above the slip plane (m); β = slope angle (degrees); γ_w = unit weight of water (= 9.8 kN/m^3); h_w = vertical height of groundwater table above the slip plane (m); and ϕ' = effective angle of internal friction of the soil material (degrees).

Figure 2.25, based on Coppin and Richards (1990), shows the main influences of vegetation on the stability of the slope segment. They can be included in the calculations of the factor of safety as follows:

$$F = \frac{(c' + c'_R) + \{[(\gamma z - \gamma_w h_v) + W] \cos^2 \beta + T \sin \theta\} \tan \phi' + T \cos \theta}{[(\gamma z + W) \sin \beta + D] \cos \beta}, \tag{2.59}$$

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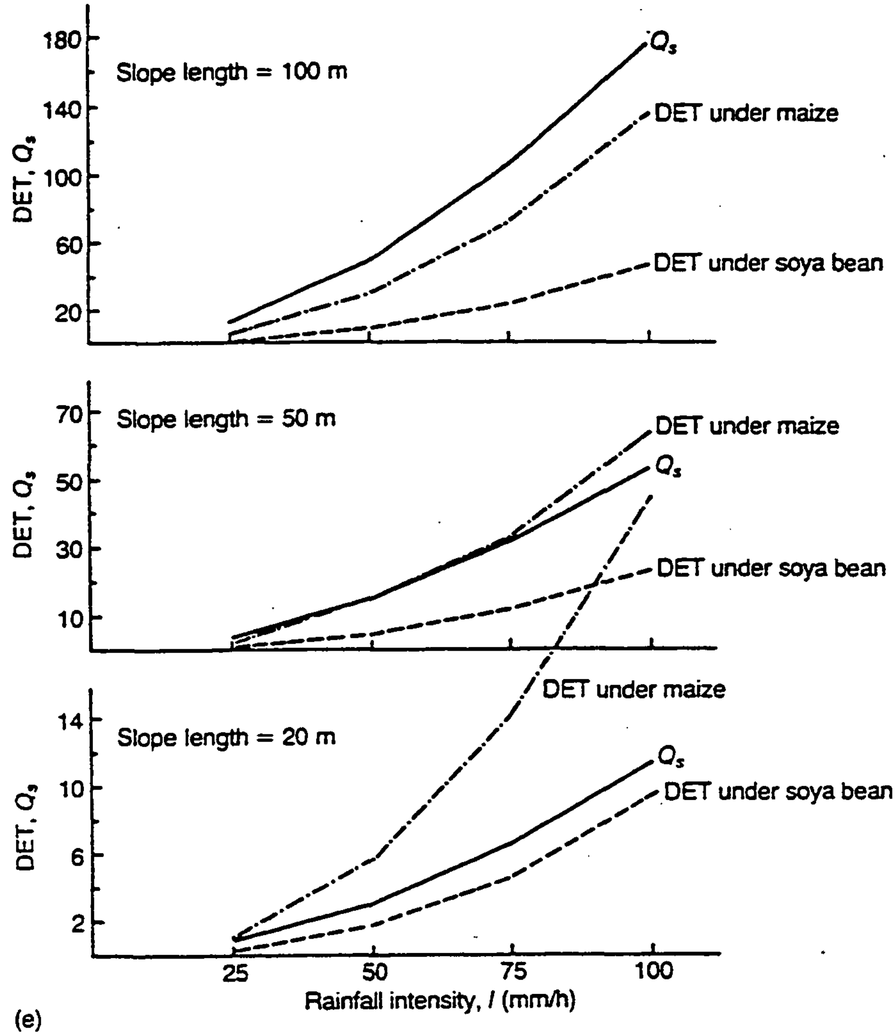

Figure 2.23 (cont.)

where c'_R = enhanced effective soil cohesion due to soil reinforcement by roots (kN/m^3); W = surcharge due to weight of the vegetation (kN/m^2); h_v = vertical height of groundwater table above the slip plane with the vegetation (m); T = tensile root force acting at the base of the slip plane (kN/m); θ = angle between roots and slip plane (degrees); and D = wind loading force parallel to the slope (kN/m).

Appendix 2.A gives the calculations for the factor of safety for a sample slope segment with and without vegetation. The calculations are purely illustrative but they show that the vegetation increases the factor of safety by 55%, assuming that the tensile strength of the roots is fully mobilized, and by 17%, if this effect (T acting over angle θ) is ignored. The greatest effects are due to the increase in cohesion through root reinforcement of the soil and to the tensile strength of the roots themselves across the potential slip surface. Although field studies of the effect of vegetation on slope stability are rare, Greenway (1987) found that the additional cohesion brought about by tree roots increased the factor of safety on wooded slopes in Hong Kong by 29%.

2.6 SALIENT PROPERTIES OF VEGETATION

The calculations presented in the previous section demonstrate that the overall effect of vegetation is the result of a balance between

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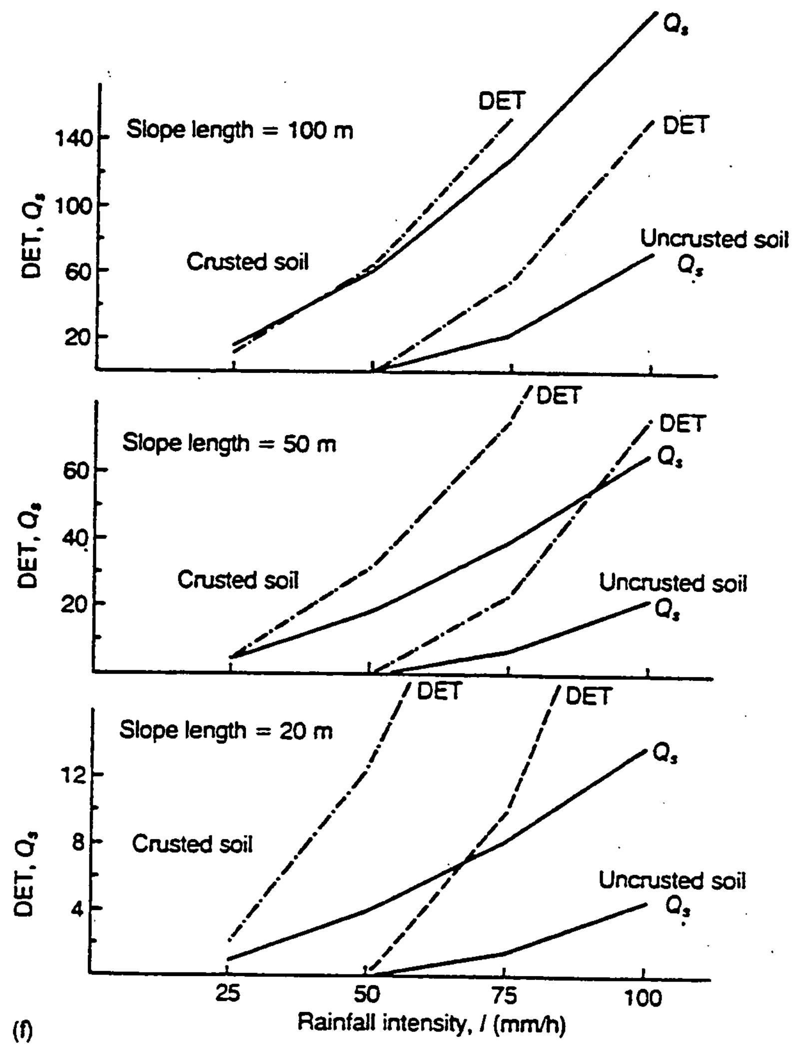

Figure 2.23 (cont.)

Table 2.16 Erosion (kg/m^2 y) in a montane forest, Java, Indonesia (after Wiersum, 1985 from table in Coster, 1938)

Cover	*Number of observations (plot years)*	*Measured erosion*	*Erosion adjusted for equivalent slope and rainfall*
Undisturbed forest	2	0.03	0.01
Trees removed	5	0.04	0.03
Undergrowth removed	4	0.06	0.05
Trees and undergrowth removed	1	0.08	0.02
Undergrowth and litter removed	10	4.32	2.61
Trees, undergrowth and litter removed	2	1.59	0.44
Shrub vegetation	4	0	0
Shrubs removed	3	0.20	0.23

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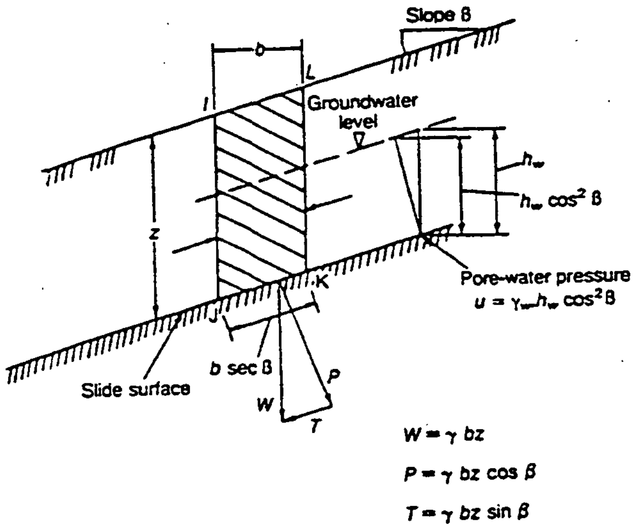

Figure 2.24 Factors involved in the infinite slope method for analysing slope stability (Notation in text.)

beneficial and adverse influences. These have been summarized by Coppin and Richards (1990), as shown in Table 2.17. The nature of the balance and, therefore, the engineering function which individual plants perform will depend upon their structure or architecture. Plants with strong tap and sinker roots will help stabilize a slope through arching and buttressing, whereas plants with a dense lateral rooting system will increase the strength of the top layer of the soil by adding to cohesion. In contrast, surface erosion processes are more strongly influenced by the above-ground growth of the vegetation. The properties of the vegetation which influence its engineering function are listed in Table 2.18, also taken from Coppin and Richards (1990). Many of these properties vary with the stage of vegetation growth and therefore alter both seasonally and, through ecological succession, over a longer time. Increasing cohesion of the soil through vegetation growth can offset long-term decreases in soil strength brought about by weathering and the fissuring and progressive softening of overconsolidated clays.

There is a close relationship between vegetation and erosion. On the one hand, vegetation, through its engineering functions, can control the amount of erosion which takes place. On the other hand, erosion can produce such a hostile and unstable environment that vegetation will not grow. The balance and competitiveness of the erosion–vegetation system has been analysed by Thornes (1988a,b, 1990) with respect to southeast Spain. He assumes that erosion limits plant growth through water and nutrient stress but that vegetation also limits erosion. Such an ecosystem may be in balance, or it may be self-reinforcing in one direction or another. For example, erosion will result in less vegetation which will produce a deteriorating water balance with less water available for plant growth and more water contributing to runoff and erosion. Alternatively, an increase in vegetation growth will lead to less erosion and a more favourable water balance for further vegetation. If grazing pressure is added to the ecosystem, a higher biomass production is necessary to keep the system in balance. Otherwise the situation may change from equilibrium to deterioration. The details of the system in southeast Spain are very complex because litter fall occurs at different times of year for different plant species and because a certain amount of the litter is removed by the runoff. One of the important effects of the ground vegetation is to keep the litter in place.

Most vegetation is self-regenerating but human and animal interference may destroy the natural cycles of plant growth. A good understanding of both natural and man-influenced ecosystems is therefore essential for analysing and predicting the engineering role of vegetation. Continuous hard grazing generally leads to loss of the vegetation cover; the plants lack sufficient leaves for photosynthesis and die, whilst damage to the growth points prevents their regeneration. These relationships are analysed theoretically in THEPROM, an erosion–productivity model being developed for rangelands in Botswana (Biot, 1990). Under special circumstances, however, the interactions may react another way. Valentin (1985) showed that, under very special ecological conditions, grassland vegetation may be improved by cattle grazing.

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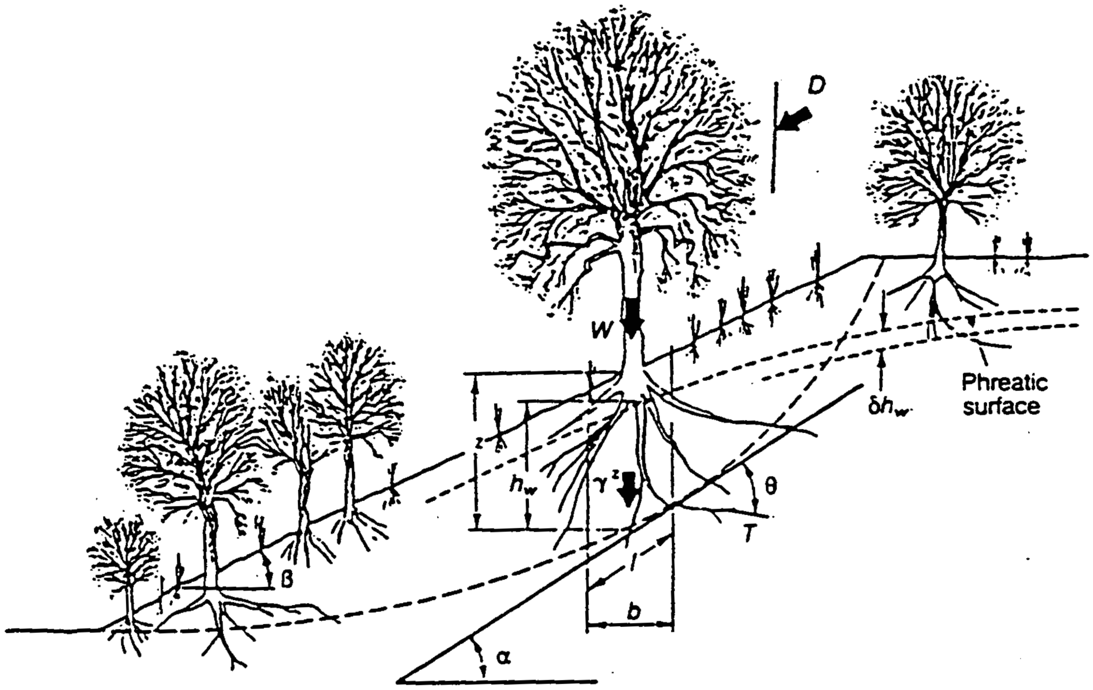

Figure 2.25 Main influences of vegetation on slope stability (after Coppin and Richards). Parameters applied in slope stability analysis:

	γ^z	Total weight of soil slice (kN/m^2)
	c', ϕ'	Effective strength parameters at slip surface
	l	Length of slip surface with slice, m (b sec α)
	u	Pore-water pressure at slip surface (kN/m^2) ($\gamma_w h_w$)
Due to vegetation	u_v	Decrease in pore-water pressure to evapotranspiration by vegetation at slip surface (kN/m^2) (2)
Due to vegetation	c_R'	Enhanced effective soil cohesion due to root matrix reinforcement by vegetation along slip surface (kN/m^2)
Due to vegetation	c_S'	Enhanced effective soil cohesion due to soil suction due to evapotranspiration by vegetation at slip surface (kN/m^2)(2)
Due to vegetation	W	Surcharge due to weight of vegetation (kN/m)
Due to vegetation	D	Wind loading force parallel to slope (kN/m)
Due to vegetation	T	Tensile root force acting at base of slice (kN/m) (assumed angle between roots and slip surface
	z	Vertical height of surface of soil layer above slip plane (m)
	h_w	Vertical height of phreatic surface or water table above slip surface (m)

Notes

1. The value of many of these parameters varies with depth and soil type.
2. In certain slope stability analyses the decrease in pore-water pressure due to vegetation (i.e. increased soil suction from evapotranspiration) is expressed as an enhanced effective soil cohesion, as distinct from a pore pressure reduction.

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Table 2.17 Beneficial and adverse effects of vegetation (from Coppin and Richards, 1990)

Hydrological effects		*Mechanical effects*	
Foliage intercepts rainfall causing:		Roots bind soil particles and permeate the soil, resulting in:	
1. absorptive and evaporative losses, reducing rainfall available for infiltration	B	1. restraint of soil movement reducing erodibility	B
2. reduction in kinetic energy of raindrops and thus erosivity	B	2. increase in shear strength through a matrix of tensile fibres	B
3. increase in drop size through leaf drip, thus increasing localized rainfall intensity	A	3. network of surface fibres creates a tensile mat effect, restraining underlying strata	B
Stems and leaves interact with flow at the ground surface, resulting in:		Roots penetrate deep strata, giving:	
1. higher depression storage and higher volume of water for infiltration	A/B	1. anchorage into firm strata, bonding soil mantle to stable subsoil or bedrock	B
2. greater roughness on the flow of air and water, reducing its velocity, but	B	2. support to up-slope soil mantle through buttressing and arching	B
3. tussocky vegetation may give high localized drag, concentrating flow and increasing velocity	A	Tall growth of trees, so that:	
Roots permeate the soil, leading to:		1. weight may surcharge the slope, increasing normal and down-slope force components	A/B
1. opening up of the surface and increasing infiltration	A	2. when exposed to wind, dynamic forces are transmitted into the ground	A
2. extraction of moisture which is lost to the atmosphere in transpiration, lowering pore-water pressure and increasing soil suction, both increasing soil strength	B	Stems and leaves cover the ground surface, so that:	
3. accentuation of dessication cracks, resulting in higher infiltration	A	1. impact of traffic is absorbed, protecting soil surface from damage	B
		2. foliage is flattened in high velocity flows, covering the soil surface and providing protection against erosive flows	B

A = adverse effect
B = beneficial effect

Table 2.18 Salient properties of vegetation and their engineering significance (from Coppin and Richar 1990)

Vegetation: Effect on	Vegetation: Influence	*Vegetation properties*: Ground cover (%)	Height	Leaf shape and length	Stem/leaf density	Stem/leaf robustness	Stem/leaf flexibility	Root depth	Root density	Root strength	Annual growth cycle	Weight
Surface competence												
	Soil detachment	●	●	●	●						●	
	Mechanical strength	●	●		●	●			●	●	●	
	Insulation	●			●						●	
	Retarding/arresting		●		●	●	●					
	Erosion	●			●						●	
Surface water regime												
	Rainfall interception	●		●	●							
	Overland flow/runoff	●			●							
	Infiltration				●			●	●			
	Subsurface drainage							●	●			
	Surface drag	●	●	●	●		●				●	
Soil water												
	Evapotranspiration			●	●				●		●	
	Soil moisture depletion leading to increased soil suction, reduced pore-water and soil weight							●			●	
Properties of soil mass												
	Root reinforcement							●	●	●	●	
	Anchorage/restraint							●	●	●		
	Arching/buttressing							●		●		
	Surface mat/net								●	●	●	
	Surcharge		●									●
	Windthrow		●		●	●		●		●	●	●
	Root wedging							●	●			
Air flow												
	Surface drag		●	●	●		●				●	
	Flow deflection		●		●	●	●				●	
	Noise attenuation		●	●	●						●	
	Suspended particulates		●		●						●	

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APPENDIX 2.A:

Estimation of the effect of vegetation on the factor of safety on a slope using the infinite slope method. Notation is given in the text.

FACTOR OF SAFETY WITHOUT VEGETATION:

$$F = \frac{c' + (\gamma z - \gamma_w h_w)\cos^2\beta \tan\phi'}{\gamma z \sin\beta \cos\beta}$$

$c' = 10\,\text{kN/m}^3$

$\gamma = 18\,\text{kN/m}^3$

$z = 1.0\,\text{m}$

$\beta = 35°$

$\phi' = 35°$

$\gamma_w = 9.8\,\text{kN/m}^3$

$h_w = 0.5\,\text{m}$

$$F = \frac{10 + \{(18 \times 1) - (9.8 \times 0.5)\} \times 0.6710 \times 0.7002}{18 \times 0.5736 \times 0.8192}$$

$$F = \frac{10 + \{18 - 4.9\} \times 0.6710 \times 0.7002}{8.4581}$$

$$F = \frac{16.1548}{8.4581}$$

$$F = 1.91$$

FACTOR OF SAFETY WITH VEGETATION

$$F = \frac{(c' + c'_R) + [\{(\gamma z - \gamma_w h_v) + W\}\cos^2\beta + T\sin\theta]\tan\phi' + T\cos\theta}{\{(\gamma z + W)\sin\beta + D\}\cos\beta}$$

$c'_R = 5\,\text{kN/m}^3$

$W = 2.5\,\text{kN/m}^3$

$D = 0.1\,\text{kN/m}^2$

$h_v = 0.4\,\text{m}$

$T = 5\,\text{kN/m}$

$\theta = 45°$

$$F = \frac{(10+5) + [\{(18-(9.8\times 0.4)) + 2.5\} \times 0.6710 + (5 \times 0.7071)] \times 0.7002 + (5 \times 0.7071)}{[\{(18 + 2.5) \times 0.5736\} + 0.1] \times 0.8192}$$

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$$F = \frac{15 + [\{((18 - 3.92) + 2.5) \times 0.6710\} + 3.5355] \times 0.7002 + 3.5355}{9.7147}$$

$$F = \frac{15 + 10.2654 + 3.5355}{9.7147}$$

$$F = \frac{28.8009}{9.7147}$$

$$F = 2.96$$

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Increase in c_R' of 5 kN/m^3	increases F by 0.59
Increase in W of 2.5 kN/m^3	decreases F by 0.04
Increase in D of 0.1 kN/m^2	decreases F by 0.02
Increase in T of 5 kN/m	increases F by 0.71
Increase in h_w of 0.1 m	increases F by 0.08

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NITRATE REDUCTION IN OXIDIZED CLAYEY TILL

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1. Introduction.

The majority of drained Danish arable soils are developed from glacial deposits of clayey till. This till covers 40-50% of the groundwater resource in Denmark. Investigations by Jørgensen (1990), Fredericia (1990), Jørgensen & Fredericia (1992), Jørgensen & Spliid (1993) and Villholth (1994) have shown that infiltration in these deposits is dominated by rapid water flow in fractures and macropores.

Previous evaluations of chemical reactivity in the tills have been based on the assumption that infiltration is homogeneously distributed through the total porosity of the material. Denitrification/nitrate reduction has been assumed to be negligible in the oxidized portion of the tills (Zeuthen et. al. 1991) due to the lack of continuous reactive organic matter throughout the till. Chemical reaction with clay minerals in the reduced zone has been suggested as a nitrate reduction process in the till (Ernstsen, 1990).

However, shallow profiles of porewater nitrate concentration decrease with depth, often reaching a concentration of zero at the base of the oxidized till zone (Dyhr-Nielsen et. al., 1991). Mineralogical evidence suggests that localized organic matter (i.e. root fragments) in the macropore structure of the oxidized zone may have a significant potential for denitrification (Jørgensen 1990). This potential would be greatest during seasonal (autumn and winter) water saturation and groundwater infiltration into the macropores/fractures.

Jørgensen (1990) has indicated that denitrification is sensitive to water saturation and the residence time of infiltration. This work concluded that traditional tile drains increase the nitrate concentration of infiltration and stressed the importance of disconnecting drains of abandoned arable land.

In this project the nitrate reduction and redox chemistry of infiltrating mobile pore water in the oxidized zone is studied by laboratory experiments using saturated columns of undisturbed clayey till. Sampling of the undisturbed columns was performed in October, 1½ months after harvest, in two excavated profiles (10x10x4m). This paper reports preliminary experimental results from a till column collected 2 m beneath arable land in Grundfør, Jylland. The project is part of the project "Transport and Reduction of Nitrate in Macropores in the Temporarily Saturated Zone and in the Groundwater Zone" funded by the Strategiske Miljøforskningsprogram.

2. Materials and methods.

The cylindrical column sampled is 0.5 m in diameter and 0.5 m high, being large enough to represent fractures and macropores in the till. During sampling the column was embedded in a fluid rubber casing which fixed the column (after hardening) in a combined mould and transport steel cylinder. Before hardening the fluid rubber enters a few millimetres of the till matrix, eliminating flow along the boundary of the outer column surface. After fixation, the column was detached from the till formation. The steel cylinder was removed after transport of the column; stabilization during installation in the laboratory was provided by creating a vaccum of - 30 to - 60 kPa inside the column. The intact till column was installed in a large triaxial cell and connected to a percolation system. In the triaxial cell the in situ pressure and temperature of the till formation were simulated in order to establish realistic physical conditions during the experiments with the columns. Figure 1 identifies the experimental apparatus for the column.

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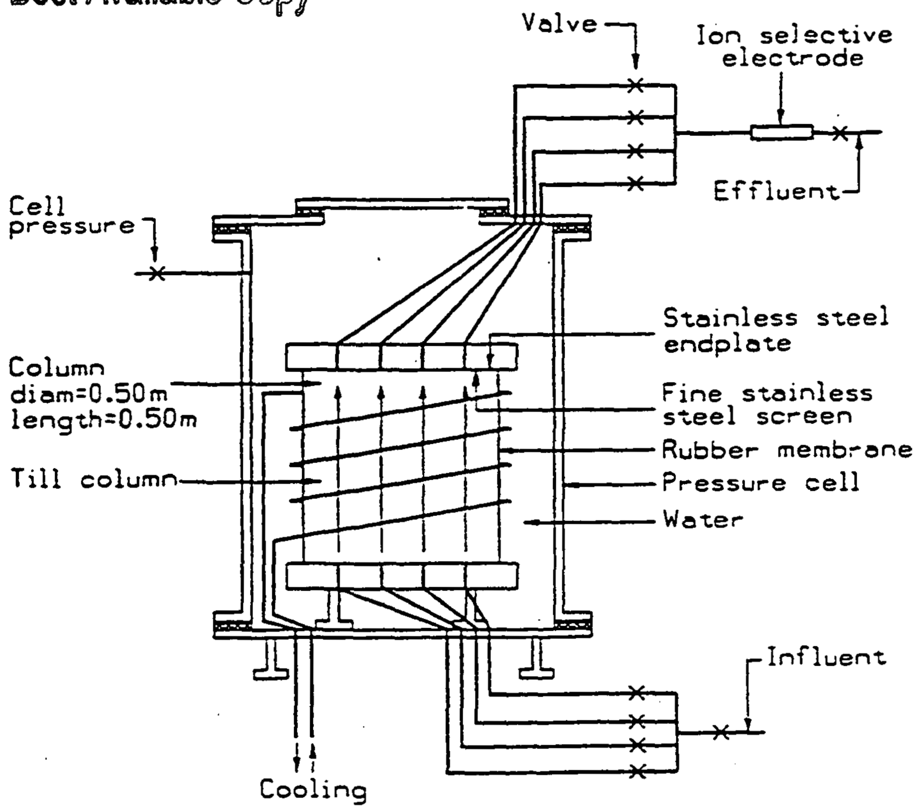

Fig.1. Experimental set-up for large undisturbed columns.

Drainage water collected at the excavation site was used as influent. It had a nitrate concentration of 85 mg/l and after addition of lime gravel a pH-value of 7.85 was measured. Bromide in the form of NaBr was added to a concentration of 0.05 mg/l. During the experiment the influent container was keept at 2°C to minimize denitrification before the start of the experiment. The influent flow was controlled by a peristaltic pump. The soil column temperature was 13°C. In glass vessels inserted into the the effluent string NO_3 and Br concentrations were measured continously with electrodes. pH and O_2 were analyzed manually on effluent samples. Samples for measuring O_2 were handled in closed measuring flasks, while samples for measuring Fe^{2+} were injected directly into evacuated acid containing blood sampling ampules. O_2 was analysed by Alsterberg modification of the Winkler method (Standard Methods for the Examination of Water and Wastewater, 16. Ed, p.416 APHA, 1985), while Fe^{2+} was measured by use of a modified 1.10-phenanthroline method described by Marzenko (Separation and Spectrofotometric Determination of Elements, p330, 1987). The flow rate of the experiments was measured continously by recording the mass of column effluent.

3. Geological profile and dual porosity structures

The site of the investigation is situated at Sandballegaard, Grundfør, which is approximately 20 km NW of Aarhus. The geology of the site consists of a clayey till plain with inclusions of local sand deposits. The glacial till overrides glaciofluvial sand and gravel; portions of the till contain "windows" of sandy meltwater deposits. The till layer has a reduced lower zone that is developed locally. The column discussed in this paper was sampled from a profile where the till layer was 3.2 m thick and does not have a reduced zone.

Mapping of the profile revealed a high density of fractures in the till. In horizontal planes, weakly pronounced fractures made a typic polygonic pattern resulting from wetting/drying and freeze/thaw cycles. The cumulative lengths of fractures measured on three excavacted planes are shown in Figure 2. As shown in the figure, the cumulative fracture length (which can be converted to an average fracture spacing) decreases with depth in the till.

In addition to the fractures, cylindrical macropores (root and worm holes) were observed in the till. The macropores range from aproximately 0.2 - 4 mm in diameter and have a density of 2-10 macropores/cm2. Both the fractures and macropores have bleached rims, especially in the lower portion of the profile, and showed pronounced redistribution of Fe(II) and Mn oxides in the matrix.

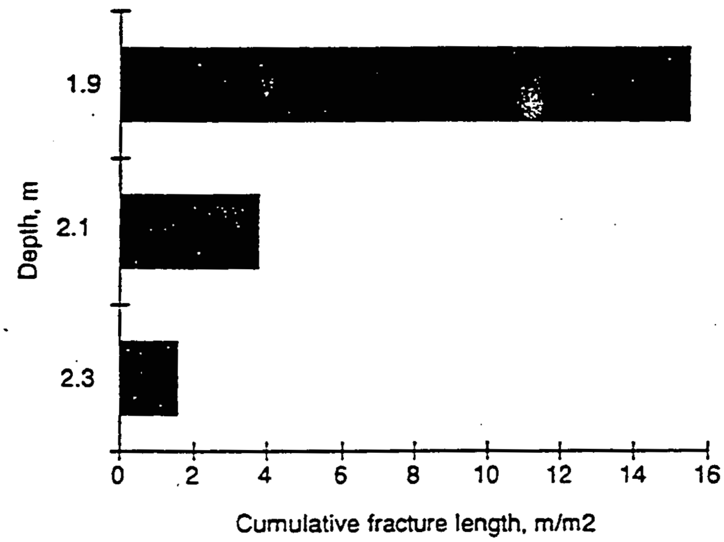

Fig. 2. Cumulative length of observed fractures from three depths in the excavated profile.

4. Geochemical indicators of denitrification.

Bleaching of fracture and macropore walls in oxidized tills is very common and associated with the occurrence of organic matter (plant roots/root fragments). An example (Jørgensen & Fredericia 1992) of the distribution of Fe(III) minerals in a bleached fracture and the surrounding matrix is shown together with the distribution of secondary Al-minerals and Fe-oxidation state in clay mineral lattices in Figure 3.

Bleaching results from reduction of secondary Fe(III) minerals in the fracture/macropore wall. According to the reaction scheme in Table 1, the reduction occurs when decomposion of root fragments (represented by CH2O in the table) causes development of anaerobic conditions in the fractures due to deficient oxygen supply from the atmosphere at water saturation in winter (Berner 1980, Jørgensen 1990).

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Reaction	ΔG^o : kJ mol^{-1} of CH_2O
1. $CH_2O + O_2 \rightarrow CO_2 + H_2O$	-475
2. $5CH_2O + 4NO^{-3} \rightarrow 2N_2 + 4HCO_3^- + CO_2 + 3H_2O$	-448
3. $CH_2O + 3CO_2 + H_2O + 2MnO_2 \rightarrow 2Mn^{++} + 4HCO_3^-$	-349
4. $CH_2O + 7CO_2 + 4Fe(OH)_3 \rightarrow 4Fe^{++} + 8HCO_3^- + 3H_2O$	-114

Table 1. Standard state free energy changes for some bacterial reactions; data for CH_2O and MnO_2 are for sucrose and fine-grained Birnessite, respectively (Berner, 1980).

From thermodynamic considerations (see Table 1), decomposition of organic compounds can be described by a succession of microbiological reactions where the more energetic reactions will dominate over other less energetic reactions. Oxygen, nitrate, manganous-oxides, and ferrous oxides are reduced in order as reations 1-4 proceed.

The distribution of the secondary Fe(III) minerals shown in Fig. 3 can be evaluated from the reactions of Table 1. According to reaction 4, the concentration of dissolved ferrous ions builds up in the fractures due to reduction of Fe(III)-minerals. From the fractures the ferrous ions diffuse into the surrounding clay matrix, where they meet dissolved oxygen. In consequence reaction 4 reverses and the ferrous ions re-oxidize and re-precipitate, building up a reddish rim which encloses the bleached macropores/fractures.

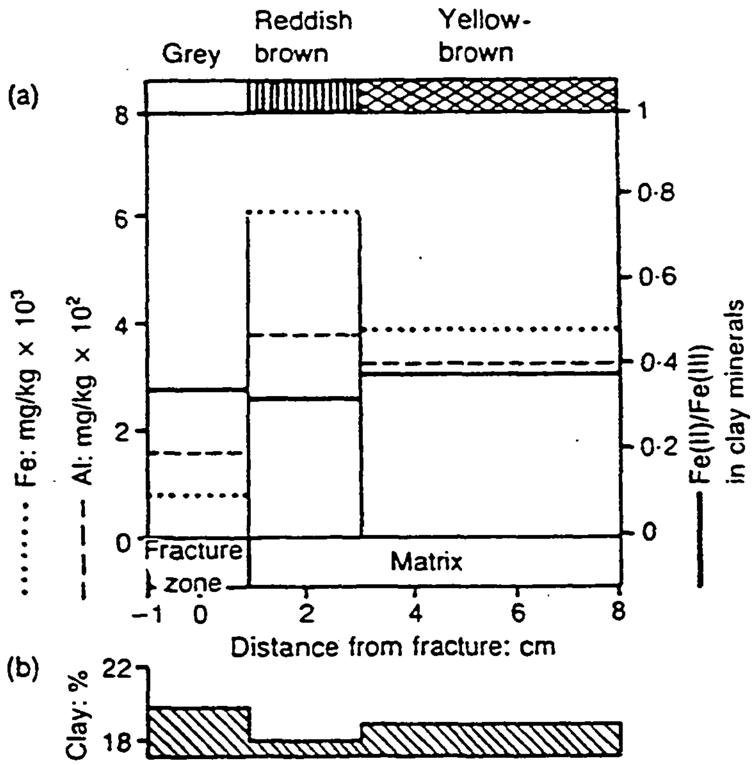

Figure 3. Chemical and sedimentological variations as a function of distance from a fracture. Analyzed samples arefrom 2.5 m depth. (a) Distribution of secondary iron and aluminum minerals and the Fe(II)/Fe(III) ratio in the lattice of clay minerals. (b) Variations in clay content.

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The mineralogical evidence suggests that fractures in the oxidized zone represent areas of enhanced microbiological activity and variable redox conditions separated by the less active clayey matrix. Thus a low redox potential and Fe(III) reduction in the fractures can occur simulataneously with aerobic conditions in the matrix. The redox succession also suggests that oxygen and nitrate should be depleted in the macropore/fracture system before reduction of Fe(III) occurs.

5. Column experiments

The objectives of the column experiments are to: 1) investigate the redox sequence (given in Table 1) using the chemistry of the column effluent, and 2) examine the relationship between denitrification and porewater residence time.

5.1. Hydraulic measurements

Measurements of specific discharge from the column (expressed as hydraulic conductivity * gradient) as a function of applied gradient are shown in Figure 4. The linear relationship is consistent with the Darcy equation and suggests that no short-circuit flow exists along the column walls. The hydraulic conductivity of the column is 5.3*10E-6 m/s which is approximatly 3-4 orders of magnitude higher than laboratory measurements from a comparable till (Foged & Wille, 1991). From the high hydraulic conductivity it is indicated that infiltration is dominated by water flow in fractures and macropores.

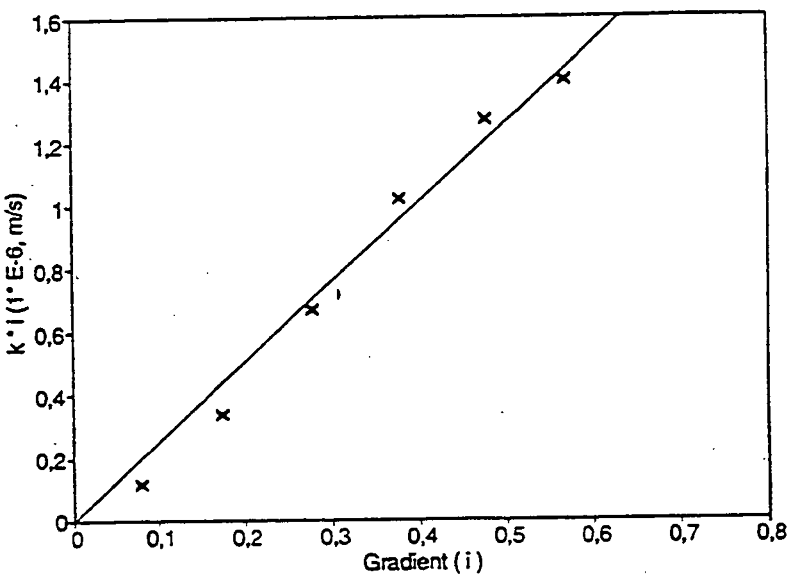

*Figure 4. Specific discharge (expressed as hydraulic conductivity * gradient) measured from the undisturbed column as a function applied hydraulic gradient.*

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5.2. Tracer experiment and numerical modelling of flow system.

Effluent concentrations versus time for a constant pulse of nonreactive (bromide) tracer in the column are shown in Figure 5. The rapid solute breakthrough indicates the dominance of transport in the fractures and macropores.

Based on the breakthrough a preliminary evaluation of the flow system with the numerical model MIKE-SHE (DHI, 1993, macropore version) confirms the dominance of transport in the macropore/-fracture system. The breakthrough curve (BTC) was successfully simulated using a total porosity of 20-30% and an effective porosity of 0.5%. This corroberates with measurements of clayey tills in other localites in Denmark (Jørgensen & Spliid, 1992, Jørgensen et.al., in.prep.)

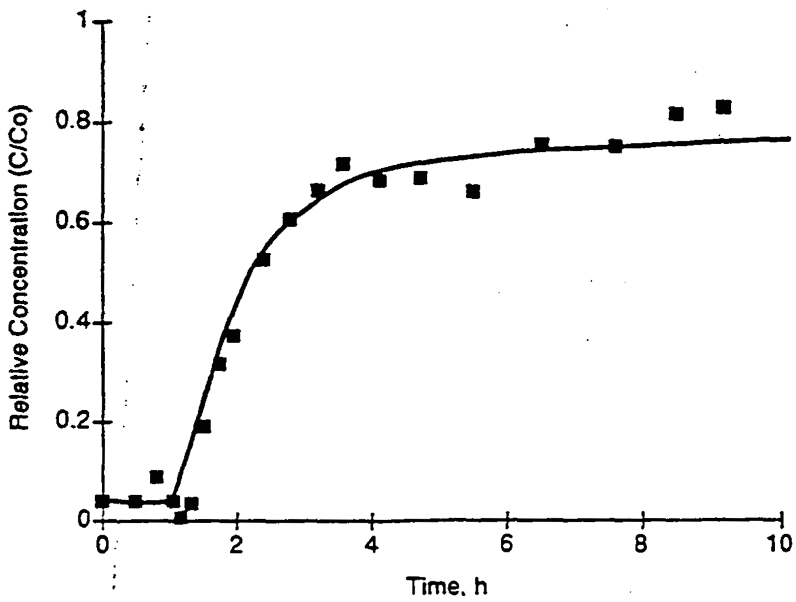

Figure 5. Observed (squares) and simulated (solid line) breakthrough curves for bromide in column effluent for a constant bromide infiltration (Q=442 ml/h).

5.3. Hydrochemical experiments.

The breakthrough curve for nitrate is compared to that for bromide in Figure 6. While bromide quickly reaches a high level of relative concentration and approches the influent concentration asymptotically, the nitrate is retarted relative to bromide and stabilizes at a significantly lower steady state concentration. As the physical and chemical transport properties of the two ions are similar, the reduction in effluent NO_3 concentration indicates rapid denitrification in the macropore/fracture system.

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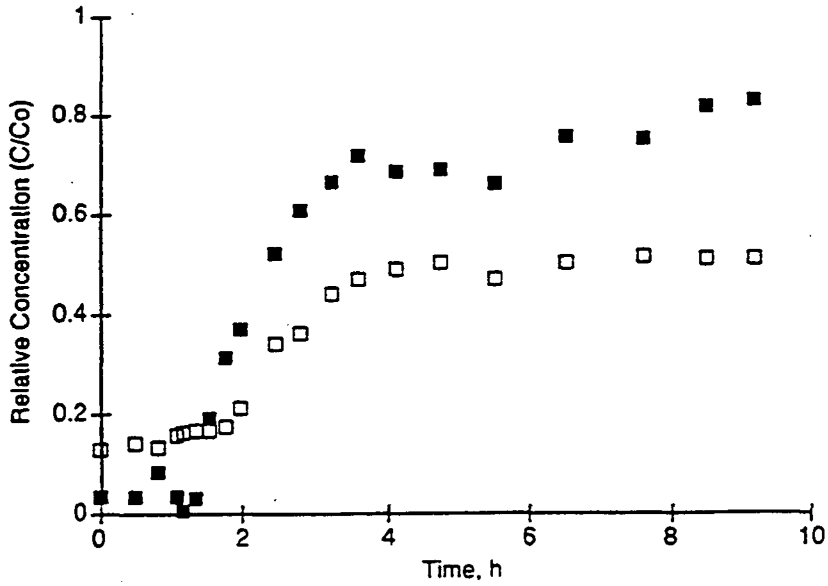

Figure 6. Breakthrough curves for nitrate (open squares) and bromide (filled squares) for a constant influent concentration (Q = 442 ml/h).

The redox sequence shown in Table 1 is investigated by analyzing O_2, NO_3 and Fe^{2+} concentrations in effluent as a function of infiltration/flow rate. A constant concentration of nitrate (85 mg/l) was applied to the column at various infiltration rates. Effluent samples were analyzed for each parameter once the nitrate concentration had stabilized for each experiment.

Predicted from the thermodynamics of a homogenous system, oxygen and nitrate should be depleted successively in the macropore system before concentrations of Fe(II) would appear in the effluent. Figure 7 shows the steady state nitrate effluent concentrations at different flow rates (porewater residence times). At an infiltration rate of 0.12 mm/h (a residence time of ~10 days) denitrification of the 85 mg/l influent concentration was complete. Increasing the infiltration rate increased the nitrate concentration in the column effluent. The maximum relative nitrate concentration tested was 0.5 at a flow rate of 1.2 mm/h. A linear regression for the data shown in Figure 7 yields the following relationship between effluent nitrate concentration and infiltration rate:

$$C_e = 41.5 * V - 2$$

where C_e = effluent nitrate concentration (mg/l) and V = infiltration rate (mm/hour)

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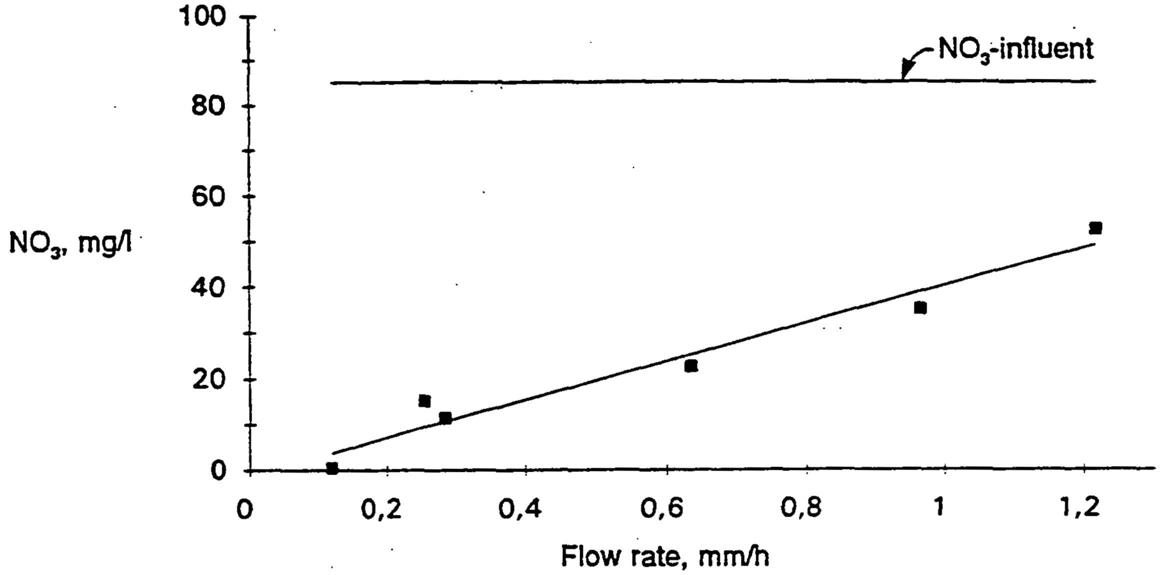

Figure 7. Steady state effluent concentrations of nitrate as a function of infiltration rate. A constant nitrate concentration is applied in the influent for each experiment. All experiments were performed at 13°C.

Figure 8 shows the effluent concentrations of O_2 and Fe^{2+} measured at steady state nitrate concentrations. The trend of these data, in addition to those shown in Figure 7, is consistent with the thermodynamic predictions of Table 1. At low infiltration rates no O_2 or NO_3 is present in present in column effluent, and dissolved ferric iron concentrations are high. As the infiltration rate increases NO_3 appears in the column effluent prior to O_2; the concentration of both constituents increases while Fe^{2+} concentrations decrease. The occurence of 0.5 mg Fe^{2+}/l at an O_2 concentration of 2.5 mg/l suggests additional complexities in the system: heterogeneities within the column with respect to redox conditions and/or the influence of Fe^{2+} complexation with dissolved organic matter.

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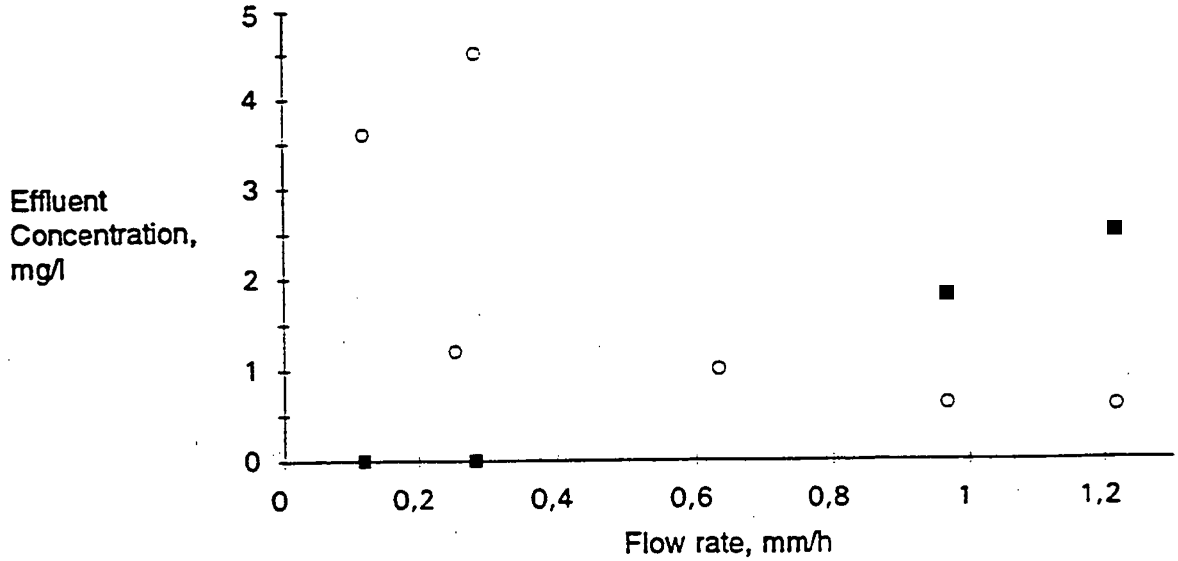

Figure 8. Effluent concentrations of O_2 (filled squares) and Fe^{2+} (open circles) in column effluent as a function of infiltration rate.

The results clearly indicate that the nitrate concentration leached through the till is very sensitive to the residence time of the porewater. This implies that permanently open drain systems of arable land contribute significantly to deterioration of the quality of infiltration to the drains and groundwater by increasing water velocities in the macropore systems.

Future research efforts will focus on quantifying the rate of denitrification as a function of porewater residence time. Additional laboratory work will be conducted and numerical simulations of the observed solute transport will be performed.

6. Conclusions

- The preliminary results obtained from an undisturbed till column sampled shows that infiltration in the clayey till is dominated by water flow in fractures and macropores.

- Saturated infiltration through the column with a constant concentration of nitrate (85 mg NO_3/l) show complete denitrification at low infiltration rates (0.12 mm/h) while approximatly 50% of the nitrate influent concentration was leached at higher infiltration rates (1.2 mm/h).

- Leaching of oxygen, nitrate and ferrous iron occurred successive in accordance with thermodynamic predictions of bacterial reactions during decay of organic matter.

- The initial experimental results suggest that traditional tile drains will increase nitrate leaching by inforcing high flow velocities and minimizing the duration of water saturated periods in the till.

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